Improving Managed Environmental Water Use: Shasta River Flow and Temperature Modeling

By

SARAH ELIZABETH NULL
B.A. (University of California, Los Angeles) 1998
M.A. (University of California, Davis) 2003

DISSERTATION

Submitted in partial satisfaction of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

Geography

in the

OFFICE OF GRADUATE STUDIES

of the

UNIVERSITY OF CALIFORNIA

DAVIS

Approved:

____

____

____

Committee in Charge
2008

Acknowledgements

I would like to express my profound gratitude to Jay Lund, my committee chair and advisor throughout graduate school. His support and guidance have been immeasurable, encouraging me to remain in graduate school to complete a PhD. Jay always has an open door, a smile, and words of wisdom whenever I need them.

I would also like to thank my committee member and boss at Watercourse Engineering, Mike Deas. Like Jay, Mike always made time in his busy schedule to answer questions, lend support, share data, as well as his considerable expertise. I will miss working at Watercourse Engineering, where I was constantly learning, and being helped to succeed in all projects to which I contributed. Thank you also to committee member, Jeff Mount, who introduced me to the tributaries of the Klamath River. I am grateful for the time all my committee members put into reading my dissertation and giving me feedback. This work would be much poorer without their insights.

Thank you to my friends in the Lund research group for providing continual sources of motivation, encouragement, and often laughter, making graduate school a fun and enriching experience. Thank you especially to Mimi Jenkins. I might have quit graduate school during my first year without her help and mentoring. Thank you also to my friends and cohort in the Geography Graduate Group.

Thank you to my friends that I have made during and prior to graduate school, whose support, senses of humor, and fun times have allowed me to thoroughly enjoy graduate school and my life during this long process. I would especially like to thank Curtis Gray, who never doubted that I would finish my dissertation. He always helped get my mind off classes and research by getting me outside (but still reminded me to keep working during my motivational lapses).

Finally, thank you to my family, whose love and support means the world to me. I would have been unable to even begin graduate school without them.

Abstract

Urban, agricultural, and industrial water users actively manage water supplies and demands to increase water use efficiency through conservation, water markets, infrastructure changes, and creative operation and management strategies. Improving water use efficiency is just beginning to be incorporated in the environmental sector to extend benefits and reduce costs. This study explores methods to improve environmental water use efficiency, as well as to reduce competition for water between environmental uses and traditional water resources. California's Shasta River is used as a case-study to evaluate restoration alternatives and efficient use of environmental water allocations. Low instream flow and high water temperature conditions are two factors limiting survival of native salmon in the Shasta River. This study examines the potential to enhance fish habitat conditions by better managing environmental instream flow and water temperature, using theoretical analysis, field monitoring, simulation, and optimization modeling. Many other potential methods to improve environmental water use efficiency exist. Results suggest that environmental water use efficiency can potentially improve environmental performance and perhaps reduce some water management conflicts. Additional research for managing environmental water use is merited. Modeling results specific to the Shasta River indicate that restoring Big Springs Creek is especially promising to enhance fish habitat in the Shasta River, cold water is necessary in the upper reaches for most restoration measures to be effective, and a combination of restoration alternatives most improves instream habitat for native salmon species in the Shasta River.

Table of Contents

Table of Figures

Table of Tables

Chapter 1: Introduction

In recent decades, urban and agricultural water agencies have actively managed their water supplies and demands to increase water use efficiency, reduce costs, and improve benefits. Improvements have been made to supply and demand infrastructure, policies, operations, and management strategies, including coordinated use of existing supplies, conjunctive use of ground and surface water, water conservation technologies, water transfers, desalination, and recycling programs (Iglesias and Blanco, 2008; Wilchfort and Lund, 1997; Yeh, 1985). This has allowed urban and agricultural agencies to stretch existing supplies to serve growing demands. Some such water use efficiency methods also could be applied to environmental water uses to increase environmental protection and restoration with limited environmental water allocations and availability.

Environmental water allocations are needed because traditional water resource development has impacted aquatic systems. Allocations for urban and agricultural water uses have historically been given priority, with environmental needs being identified more recently. A balance must be made between water resource management for traditional water uses, such as urban, agricultural, and industrial supplies, and newer water resources such as environmental and recreational uses. Improving water use efficiency in all water use sectors may reduce competition for limited water supplies.

In recent decades in the U.S., water has been allocated for environmental uses, often through legislative or regulatory processes (CDWR, 1998). Examples of environmental water uses include instream flow dedications, pulse flood releases (for seed recruitment, river channel geomorphology, fish migrations, etc.), wetland mitigation, and California's Bay-Delta outflows. Environmental water typically also includes undeveloped water, such as uncontrolled flood releases or water evapotranspirated by native vegetation (CDWR, 1998). However, this research focuses on managed environmental water, and thus necessarily concerns itself with environmental water allocations that can be controlled, quantified, and manipulated for environmental enhancement. Managed environmental water is water specifically allocated to improve or enhance environmental conditions. Environmental conditions can be single species, habitats, rivers, lakes, riparian zones, ecosystem functions, processes, or ecosystem elements, such as food webs.

Managed environmental water use efficiency (WUE) uses creative operations and management strategies to maximize environmental benefit for a given amount of water (Begley et al., 2006; Deason et al., 2004; Lankford, 2003). Here, emphasis is on increasing environmental benefit from allocated water, rather than decreasing environmental water allocations to maintain current protection. Environmental WUE is important because the environmental water sector must likely consider and improve efficiency to continue increasing environmental benefits with limited water availability, as urban and agricultural water sectors have improved efficiency to stretch water supplies.

Despite good intentions, a considerable allocation of water, laws, and protective legislation, many aquatic species and habitats have disappeared in recent decades because water allocations and restoration efforts have been unable to keep pace with environmental degradation that has occurred in the previous century (Lackey, 1999; Nehlsen, 1991). Although water is being 'invested' in the environment, there has often been little or no obvious improvement in habitat quality or ecosystem health. Current uses of environmental water alone often cannot meet goals of protecting or enhancing riparian and instream habitats, and ecosystem function. In recent decades, much effort has focused on securing additional water supplies for the environment. While worthwhile in the past, much of California's developable water has already been allocated, so additional large quantities of water for environmental use are unlikely. Managed environmental WUE is now likely to be effective to improve environmental benefits with existing supplies, as is the case with urban and agricultural water users.

Environmental WUE may have potential to improve environmental conditions in a variety of ways, such as environmental water banks, levee setbacks, combining environmental uses with traditional water uses, minimum instream flows, pulse flow releases, dam removal, or temperature control devices on reservoirs. These examples are further defined and discussed in the following chapter. This research focuses on water use in the American West, although the ideas and models developed here are applicable to any regions with environmental water conflicts.

In subsequent chapters, the Shasta River, a tributary to the Klamath River in Northern California, is used as a case-study to examine the potential of environmental water use efficiency to improve instream conditions and maximize fish habitat and population. Coho fish habitat and population are the only environmental water uses considered, and instream flow and water temperature are the primary criteria used to describe and rank habitat. The hypothesis is that potential exists to improve the health and status of native fish populations by managing limited environmental water and monetary allocations more creatively and effectively. As with water use efficiency in other sectors, this may involve changes in infrastructure, technology, and operations regarding both supply and demand.

Considerable research and analysis have been undertaken in the Shasta River basin (CDFG, 1997; Abbott, 2002; Deas et al., 2003; NRC, 2004; Deas et al., 2004; Geisler, 2005; Jeffres et al., 2008). Previous studies have helped provide direction and identify a clear understanding of local issues and problems. However, most studies were developed in response to specific questions. A comprehensive study of the Shasta River Valley is needed, and is undertaken here to understand how the system works as a whole, evaluate potential restoration alternatives, and analyze how basin-wide management approaches or combinations of approaches may improve conditions for coho salmon.

This study relates restoration costs with improvement to instream habitat provided by restoration alternatives. Theoretical analysis, field monitoring, simulation modeling, and optimization modeling are used to highlight promising restoration alternatives and increase understanding regarding salmon habitat benefits from increasing instream flow, and benefits from water temperature improvements. This dissertation has six additional chapters.

- Chapter 2 develops the concept of environmental water use efficiency, and discusses it from a theoretical standpoint. Two illustrative optimization models are developed. The first evaluates benefits to fish from increasing flow and decreasing water temperature, and the second presents possible benefits of specializing rivers for environmental and urban water uses.

- Chapter 3 provides a background on historical and current conditions in California's Shasta River, which is used as a case-study in subsequent chapters.

Factors that limit salmon production in the Shasta River are discussed, with previous research and data gaps noted.

- Chapter 4 summarizes methods and observations from field monitoring, including longitudinal water temperature analysis, exploratory thermal diversity probing, winter water temperature sampling, and tailwater return monitoring. Field data are analyzed to understand fine-scale variability not evident in simulation and optimization models.

- Yearlong model simulations of the Shasta River are developed and used to compare habitat improvement alternatives for the Shasta River in chapter 5. Alternatives include current conditions, increased riparian shading, decreased surface water diversions, tailwater return management alternatives, restored spring complexes, scheduling changes to Dwinnell Dam, removal of Dwinnell Dam, and unimpaired conditions.

- Chapter 6 uses optimization modeling to highlight how improvements to instream flow or water temperature affect habitat capacity of one fish species, and estimates the cost of restoration for each alternative. The tradeoff between fish production and the restoration costs highlight the most promising methods of improving fish habitat while preserving the existing societal values in the Shasta Valley.

- The final chapter provides a discussion of the major findings from this dissertation and outlines future and ongoing work.

References

Abbott, A.G.P. 2002. The Effect of Riparian Vegetation on Stream Temperature in the Shasta River. Master's thesis. University of California, Davis. Department of Civil and Environmental Engineering.

Begley, L., K. Edwards, S. Heard, L. Walling. 2006. Managed Environmental Water Use Efficiency: Evidence in Practice. Prepared for CDWR. U.C. Berkeley Goldman School of Public Policy. Berkeley, CA.

CDFG (CA Dept. of Fish and Game). 1997. A biological needs assessment for anadromous fish in the Shasta River, Siskiyou County, California. CA Dept. of Fish and Game, Northern California-North Coast Region. Redding, CA.

CDWR (CA. Dept. of Water Resources). 1998. California Water Plan Update Bulletin 160-98. Sacramento, CA. Available online: http://www.waterplan.water.ca.gov/previous/b160-98/TOC.cfm. Accessed 9/2008.

Deas, M.L., A. Abbott, A. Bale. 2003. Shasta River Flow and Temperature Modeling Project. Prepared for Watercourse Engineering, Inc. Sponsored by the Shasta Valley Resource Conservation District with funding by CDFG from the CA. Coastal Salmon Restoration Program. Napa, CA.

Deas, M.L., P.B. Moyle, J. Mount, J.R. Lund, C.L. Lowney, S. Tanaka. 2004. Priority Actions for Restoration of the Shasta River – Technical Report. Prepared for The Nature Conservancy. June 11.

Deason, J., Fast, J., Schroeer, L., Turner, B., van Staveren, R. 2004. Considering Water Use Efficiency for the Environmental Sector. Prepared for CDWR, Statewide Water Planning Branch. UC Berkeley, CA.

Geisler, L. 2005. Shasta River Flow, Temperature, and Dissolved Oxygen Model Calibration Technical Report. Prepared for ICE, UC Davis and NCRWQCB. UC Davis and Watercourse Engineering, Inc. Available online: http://www.waterboards.ca.gov/northcoast/programs/tmdl/shasta/060707/20AppendixDShastaRiverFlowTemperatureandDissolvedOxygenModelCalibrationTechnicalReport.pdf. Accessed 5/2007.

Iglesias, E., Blanco, M. 2008. New directions in water resources management: The role of water pricing policies. Water Resour. Res., 44, W06417.

Jeffres, C., E. Buckland, J. Kiernan, A. King, A. Nichols, S. Null, J. Mount, P. Moyle, M. Deas,. 2008. Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California – Water Year 2007. Prepared for the California Nature Conservancy by U.C Davis Center for Watershed Sciences and Watercourse Engineering, Inc.

Lackey, R.T. 1999. Salmon Policy: science, society, restoration, and reality. Environmental Science and Policy. 2 (99): 369-379.

Lankford, B.A. 2003. Environmental Water Requirements: A Demand Management Perspective. Journal of the Chartered Institution of Water and Environmental Management. 17(1): 19-22.

Nehlsen, W., Williams, J.E., Lichatowich, J.A. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. Fisheries. 16(2):4-21.

National Resources Council (NRC). 2004. Endangered and Threatened Fishes in the Klamath River Basin: Causes of Decline and Strategies for Recovery. Committee on Endangered and Threatened Fishes in the Klamath River Basin, National Research Council. The National Academy Press.

Wilchfort, O., J.R. Lund. 1997. Shortage Management Modeling for Urban Water Supply Systems. Journal of Water Resources Planning and Management. 123(4):250-258.

Yeh, W. W-G. 1985. Reservoir Management and Operations Models: A State-of-the-Art Review. Water Resources Research. 21(12):1797-1818.

Chapter 2: Environmental Water Use Efficiency Theory

This chapter describes environmental WUE and its potential as a management strategy. A literature review on environmental WUE follows, with examples of environmental WUE, novel methods of environmental water management, and how the idea of restoration efficiency relates to environmental WUE. Summaries of successful and unsuccessful environmental WUE examples are included. Focus is on California, although environmental WUE could apply to any location where water scarcity is a problem and greater environmental protection is desired for aquatic systems or other habitats.

Environmental WUE potential is illustrated with two optimization models. The first model optimizes fish as indicators of environmental quality, where changes to instream flow and water temperature can improve fish habitat. This provides a method to evaluate relative environmental improvement (i.e., fish habitat) among different restoration alternatives. The model is illustrated with a series of related examples. The second model represents competing water uses over two rivers, and shows specializing rivers for fish production and other economic uses, such as water supply, may be optimal. The chapter concludes with a discussion of limitations of environmental WUE.

Environmental Water Use Efficiency

In recent decades, effort has focused on securing additional water supplies for the environment, usually as minimum instream flows, which are often mandated on rivers throughout California and the nation. Environmental water is required because humans have extensively developed water resources for urban, agricultural, and industrial uses. Prior to this water resource development, all water was used for environmental purposes. With the environmental movement in the 1960's, some developed water was allocated back to the environment for protection and enhancement of aquatic species and habitats. However, with growing urban populations and continued agricultural demands, large new allocations of environmental water will be increasingly difficult to obtain (Figure 1).

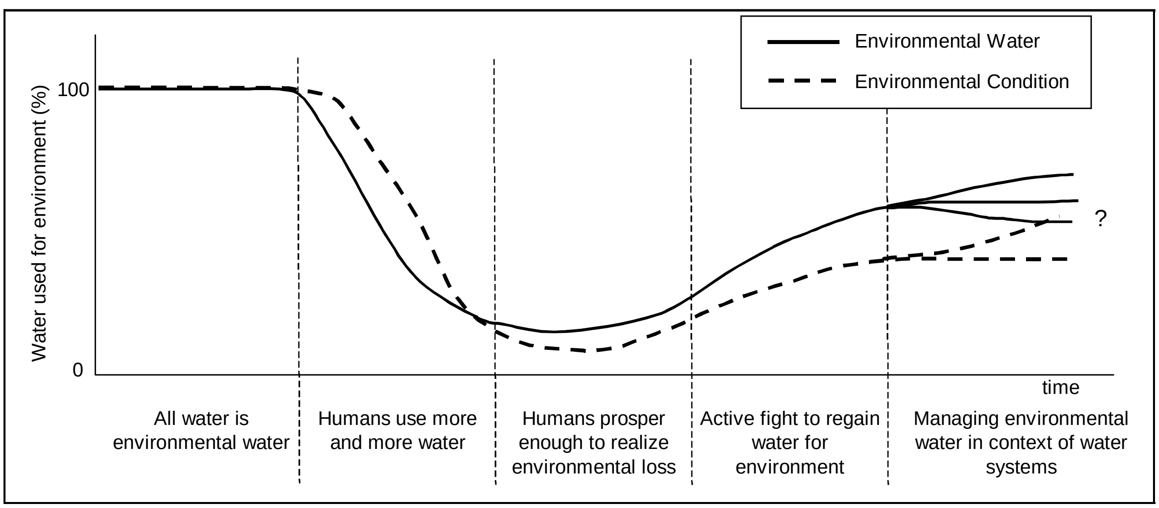


Figure 1. Conceptualized history of environmental water use

This research explores methods to quantify environmental benefit from a given amount of water dedicated to environmental uses, with a focus on improving environmental benefits. Like urban and agricultural WUE, options sometimes exist to improve environmental efficiency, such as environmental water banks, scheduling minimum instream flows to mimic natural hydrographs or fish migrations, substituting water of different qualities with other users (assuming some other users are less sensitive to water quality), and quantifying the benefit of environmental water to ensure more effective use.

Environmental WUE is not yet widely accepted as an environmental management strategy. Proponents believe it may be a tool to improve environmental protection and enhancement, ultimately saving money, time, water, and species. Skeptics believe it is not useful for improving environmental protection because it is difficult to quantify environmental improvements, it could be misused as a strategy to reduce environmental water allocations, and long term restoration commitments must be made (Begley et al., 2006). Little has been written on environmental WUE, although a report by Deason et al. (2004) was included in CDWR's 2005 California Water Plan Update (CDWR, 2005) and the idea has been discussed among water managers in California (Begley et al., 2006; Deason et al., 2004). Table 1 provides a summary of environmental WUE examples, and the following paragraphs discuss select examples.

Table 1. Strategies for environmental water use and protection through time

Option	Examples and Explanation	References
Past Strategies		
Infrastructure Changes		
Fish hatcheries	Throughout U.S.	Kauffman et al., 1997
Engineered river channels and environments	Dutch coastal engineering; bank stabilization, instream structures, artificial riffles	Kauffman et al., 1997; Disco, 2002; Brown and Pasternack, 2008
Habitat Enhancement		
Dilute, retain, or treat pollutants	Required for most urban discharge, i.e.: Iron Mountain Mine (CA), before 1988,	US EPA, 2006
Present Strategies		
Operations and Management		
Environmental water transfers/banks	EWA (CA), EWP (CA), proposed at Klamath River (OR, CA)	Winternitz, 2001; Jones and Stokes, 2003; Hollinshead, 2005; Burke et al., 2004
Mitigation banking	Kimball Island (CA), Campbell Ranch (CA)	US EPA, 1995
Managed wetland	Manage wetland area, volume, depth for water supply and quality; Design of wetlands to benefit wildlife (common)	Lankford, 2003; Kelley, Jr., et al., 1993
Optimize restoration level	Mono Lake (CA), Deckers Creek (WV)	Hart, 1996; Collins et al., 2005
Combine environmental use with other uses	Yolo Bypass/Yolo Basin Wetlands (CA); San Joaquin River release reused for water supply (CA)	Sommer et al., 2001; Sommer et al., 2003
Water Scheduling		
Minimum instream flows	Common. California examples: Trinity, Sacramento, American, Feather, San Joaquin, Tuolumne Rivers (CA), Putah Creek (CA), CVPIA (CA)	Gillilan and Brown, 1997
Pulse flows: migration cues, geomorphology…	Proposed on Yuba, American, Mokelumne, Merced, Tuolumne, San Joaquin Rivers (CA)	AFRP, 2001; Richter et al., 1996; Poff et al., 1997
Channel wetting: conjunctive use	Cosumnes River (CA)	Fleckenstein et al., 2001
Infrastructure Changes		
Dam removal/removal of migration barriers	Battle Creek dam removal (CA), Red Bluff Diversion Dam fish passage (CA)	Kier Associates, 1999; USBR website
Temperature control curtains/devices	Shasta Dam (CA), Whiskeytown and Lewiston Reservoirs (CA), proposed at Glen Canyon Dam (AZ)	Vermeyen, 1997; GCMRC, 1999
New water treatment facilities	Iron Mountain Mine (CA), 1988	US EPA, 2006
Habitat Enhancement		
Compatible water and environmental uses	Sacramento Valley waterfowl and rice cultivation	Shuford, 1998; Heitmeyer, 1989
Riparian shading	Sacramento River (CA), Shasta River (CA)	Lowney, 2000; Deas, 2000
Levee setbacks	Maximize flooded habitat; Provide habitat diversity	Kelley, Jr., et al., 1993
Dust mitigation	Owens Dry Lakebed (CA)	Anderson, 2006
Possible Future Strategies		
River specialization	Specialize systems for environmental or other economic water uses; Battle Creek (CA) dam removal	Mar, 1981, 1998; Bay-Delta Authority, 2006
Weed management	Replace high ET species with lower ET species, i.e.: tamarisk, yellow starthistle	Zavaleta, 2000; Davenport et al., 1982

The most widely discussed example of environmental WUE in California is the Bay Delta Authority's Environmental Water Account (EWA) for San Francisco's Bay Delta. The EWA buys water from willing sellers, providing carriage water to balance delta salinity and augment flows for migratory fish species (Hollinshead, 2005; Winternitz, 2001). The EWA demonstrates that environmental water transfers can sometimes improve water use efficiency and increase environmental protection. Environmental water banks also are being considered in the Klamath River Basin to stretch limited environmental water supplies because they allow for more control to move water when and where it is needed (Burke et al. 2004).

The Bay Delta Authority has a program similar to the EWA called the Environmental Water Program, which can buy water for instream uses upstream of the Bay Delta. To date, no water transfers have been made, although funding or other support may exist (Jones & Stokes, 2003). This program was intended to enhance upstream habitat areas using methods similar to the EWA.

In addition to environmental water transfers, other operational options exist to improve environmental WUE. Controlling the area, depth, velocity, and residence time of engineered wetlands through water timing, placement, and scheduling decisions may reduce environmental water demand. Man-made wetlands, or other engineered habitats should be hydrologically, ecologically, and economically efficient (Kelley Jr. et al., 1993; Lankford, 2003). Lankford is one of only a few authors who mention environmental WUE explicitly, arguing the environment should be included in water efficiency and productivity analysis, as are other water users.

In California and other regions with competition for water among environmental and economic interests and water delivery systems already near capacity, native plants and wildlife are in the difficult position of competing with other interests for increasingly expensive water. In these instances, environmental uses can sometimes be combined with traditional water uses. This occurred with California's Yolo Bypass off-stream flood channel, beginning with the realization that the bypass was remarkably productive for local fish species and migratory birds (Sommer, 2001). This resulted in undeveloped water providing habitat for migratory birds and fish in the Yolo Wildlife Area (Yolo Basin Foundation, 2006). Also as wetlands have disappeared in the state (and throughout the nation), Sacramento Valley rice fields have been identified as productive waterfowl and shorebird habitat, and cultivation is being modified to support these species (Shuford et al., 1998; Heitmeyer et al., 1989).

Throughout California and the western U.S., minimum instream flows have been used to ensure environmental water allocations. In some river systems, such as rivers throughout California's Central Valley, instream flow scheduling has been altered to mimic a more natural hydrograph, with fluctuations in flow magnitude, duration, flashiness, timing, and frequency (AFRP, 2001; Richter et al., 1996). Hydrologic variability is ecologically and geomorphically important. Pulse flows can maintain river channel integrity, improve seedling recruitment, and cue migratory species (Poff et al., 1997). For efficient environmental use, ecologists and water managers should test how much water is needed for desired goals (to mobilize gravels, flush sediment, cue migrating fish, etc.) and release only the required water, possibly preserving water for additional environmental functions. Altering instream flow schedules is a technically easy method of allocating environmental water to be more environmentally effective (Poff et al., 1997), although implementation can be difficult politically and legally, as occurred at California's Putah Creek (Fell, 2000).

California has extensively developed its water resources; thus changes to infrastructure can sometimes improve environmental WUE. Recently, removing dams has become more common to restore ecological connectivity or aesthetic values, especially where re-operation of other dams in series may provide nearly the same benefits (Null and Lund, 2006). Battle Creek, at tributary to California's Sacramento River, is a prime example. Five small hydroelectric dams are being removed to restore anadromous fish passage, and flow will be increased on three additional hydroelectric dams to minimize hydropower losses (Bay Delta Authority, 2006). Renovating and re-operating aging or outdated infrastructure may improve overall efficiency, allowing greater water dedications for environmental uses.

For certain species, cold water has a higher environmental value than warm water during summer periods (or vice versa). Water temperature control devices or curtains offer another method to provide water temperatures that support fish without additional water dedications. Temperature control devices have been used on Shasta, Whiskeytown, and Lewiston Reservoirs in Northern California to draw cold water from depth rather than warm surface water (Vermeyen, 1997).

Finally, constructing water treatment facilities instead of diluting pollution often results in water savings. Prior to 1988, dilution of acid mine drainage was used at Iron Mountain Mine. In 1988 the US EPA built a treatment facility to remove metals and other contaminants (US EPA, 2006). New treatment facilities require an investment of capital, but may be preferable to dilution when water is scarce.

Not all projects that have tried to maintain environmental value while decreasing water or other resources have been successful; fish hatcheries are such an example. In the early 1900's when rivers throughout the American West were developed for water supply, flood protection, and hydropower, fish hatcheries were offered as a solution to the loss of natural fish habitat. It was thought that economically important fish stocks could be maintained while river systems were developed for other uses. Hatchery fish were later discovered to be poor substitutes for wild salmon and trout, with problems associated with altered run timing, loss of genetic diversity, competition with wild fish, and disease (Nehlsen et al., 1991; Lackey, 1999). The improved operations of existing hatcheries might be a promising environmental WUE option; however, this example emphasizes the need for an adaptive management framework when implementing environmental WUE strategies.

In the future, new strategies to improve environmental WUE will likely be developed. Removing exotic vegetation with high evapotranspiration (ET) rates and specializing rivers (dedicating some rivers for environmental protections while extensively developing other rivers) are two promising approaches for increasing instream flow. Preliminary modeling and lysimeter studies suggest replacing invasive weeds with more drought tolerant species may be promising for managing environmental water in upland and riparian areas if invasive species have high ET rates. Tamarisk and yellow starthistle are examples of two such species in the American West (Null, unpublished data, 2007; Zavaleta, 2000; Davenport et al., 1982). Replacing these plants with species with lower ET rates may save water, which could then be used for other environmental water uses. Likewise, specializing rivers for different economic or environmental uses may become more widespread in the future. This will be discussed extensively in later sections.

Restoration Efficiency

Related to the idea of environmental WUE, is the concept of restoration efficiency, meaning the money dedicated to restoration, as well as the water, should accomplish as much environmental benefit as possible. While there has been little research on environmental WUE specifically, the idea of restoration efficiency is not new. Growing concern in the media is indicative of fears that restoration funds are not spent as efficiently as possible (Cornwall, 2005; Anderson, 2006; Harden, 2004). Not only should environmental water and monetary dedications be accounted for, but so should the success of programs at meeting restoration goals. Environmental programs must be more accountable in terms of setting goals, quantifying improvements, and comparing alternative restoration options to know if environmental water dedications and restoration funds are being used efficiently. Accountability of water use and economic costs are imperative in the urban and agricultural water sectors and have led to much greater urban and agricultural benefits from limited water use. It is likely that environmental protection would improve if the environmental sector better related benefits of restoration programs with water use and associated costs.

A common technique for assessing restoration efficiency is to evaluate a desired level of restoration. From an environmental perspective, it would be ideal to restore degraded areas to their levels before human intervention. But restoration costs, invasive species, and competing water uses often make this infeasible and unrealistic. In these instances, choosing a level of restoration to meet desired goals is needed. Restoration of Mono Lake is an example where there is not enough water to meet all environmental and human demands. Mono Lake is currently being restored, but not to the historic lake level prior to construction of the L.A. Aqueduct (6,417ft [1,956m] above sea level). Rather, it is a compromise with other economic uses such as water supply and hydropower generation. The lake level will increase to 6,392 ft (1,948m) above sea level, where it will be maintained. This level prevents the naturally saline lake from becoming too salty to support aquatic life, reduces dust by limiting exposure of the lakebed, ensures the lake's islands do not become connected to land, and preserves scenic quality of the region (Hart, 1996). Once the lake rises to this prescribed level, L.A. can divert excess water.

A recent study for Decker's Creek in West Virginia also evaluated restoration level. The public was surveyed as to whether they want (and would pay for) the creek to be restored for aesthetic value, swimming and wading, a put and take fishery, or a self-sustaining fishery. Restoration cost estimates were included for each restoration decision (Collins et al., 2005).

Mitigation banking is another approach that incorporates elements of restoration efficiency. Mitigation banking is wetland restoration or enhancement to compensate for wetland losses from development at another site, where restoration at the developed site would not be beneficial, or is otherwise infeasible. Mitigation banking has been both cheaper and more successful than traditional wetland restoration (Environmental Defense, 1999). Mitigation banks can be created for endangered species or key habitat regions can be restored, providing ecological connectivity which is more valuable than isolated pockets of habitat (Williams et al., 2003).

In some cases, water has been dedicated for environmental use; however, it is unclear how best to use that water to enhance environmental conditions. This has occurred with California's Central Valley Project Improvement Act (CVPIA). The CVPIA is unique because environmental water allocations were specifically quantified, but methods of determining the best use for environmental water are lacking which slows restoration (Sunding, 2003). Environmental WUE and restoration efficiency strategies can provide a framework for this type of practical management problem.

Water Use Efficiency for Fish Habitat and Production

The remainder of this chapter focuses on water management improvements for fish habitat and production; although fish habitat enhancement is just one example for which environmental water use efficiency may be environmentally and economically beneficial. The potential of environmental WUE to improve instream conditions for aquatic species is explored using systems analysis. This section begins with an introduction to systems analysis and optimization because theoretical environmental WUE examples are illustrated using simple optimization formulations. First a steady state optimization model with one fish species introduces the concept of optimizing flow and water temperature for fish habitat capacity. In the next formulation, detail is added as the egg life history age class cannot move freely between reaches. Next, a time component is added so that habitat conditions are changeable by the chosen timestep.

These ideas are tested with a simple steady state optimization model created in Microsoft Excel, which estimates different rates of heating in a mainstem river reach and an irrigation channel, assessing how various return flow locations alter fish habitat in the mainstem river. This illustrates potential applications of environmental WUE, introducing concepts and methods that are further explored in Chapter 6. The following section examines the potential of optimizing river systems by use in a simple system, specializing one river for fish habitat, and another river for urban water use. This chapter ends with conclusions on the concepts introduced here.

Systems Analysis and Optimization Background

Systems analysis is an interdisciplinary approach for analyzing complex problems with interacting parts, and identifying the best course of action than might otherwise have been found (Labadie, 2004). Systems analysis is related to operations research, which uses mathematics, statistics, optimization, simulation, and decision analysis to find optimal solutions to intricate problems. The scenarios presented below are simplified examples using systems analysis and optimization. Optimization is an approach to systems analysis that explicitly seeks the 'best' solution to a problem within constraints. An objective function expresses the goal of the model, which is maximized or minimized to arrive at an optimal solution. Constraints define the feasible region. The objective function and constraints are mathematical functions of decision variables and parameters. Decision variables are values which are changeable in the model representing management decisions and parameters are given (Hillier and Lieberman, 1967; Cohon, 1978). The example models demonstrate the theoretical value of environmental WUE for improving environmental performance.

Application of systems analysis in water resources has focused primarily on simulation and optimization of human water uses including urban and agricultural water reliability, flood control, hydropower generation, and to a lesser extent, recreation uses (Cardwell, 1996). Instream flow demands are typically modeled as constraints removing them from decision-making. Few modeling studies have included environmental objectives with traditional human-based objectives; although Cardwell (1996) used multi-objective optimization to improve water reliability and fish habitat in a simple reservoir-stream system. His model is similar to the optimization in the following examples (although formulation differs between the two models and examples here include water temperature). Interestingly, Cardwell's example was intended as a planning model to help with FERC relicensing, while the models here are used as theoretical environmental WUE examples.

In this section, all models optimize out-migrating fish under variable habitat conditions. Fish carrying capacity in each reach can be altered only by changing instream flow and water temperature. Other potential stressors such as biotic interactions, bioenergetics, disease, abundance of food, and habitat complexity are all assumed to be ideal for fish survival, and are constant. Fish have five different age classes (in-migrants, eggs, alevin, juvenile rearing, and out-migrants). Here flow and temperature requirements do not change among age classes, although in reality requirements vary (Table 2). Rivers are represented by a series of connected reaches (Figure 2). Instream flow and water temperature are constant within each reach, but variable between reaches. Models are loosely based on habitat requirements and life history of coho salmon in California's Shasta River, although these are primarily proof of concept examples.

Table 2. Optimal water temperature requirements by age class (Deas et al., 2004; NRC, 2004; DFG, 2002)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
In-migration											10-12 C	
Eggs											4-9 C	
Alevin			4 - 13 C									
Juvenile Rearing						12-14 C						
Out-migration			8 - 16 C									

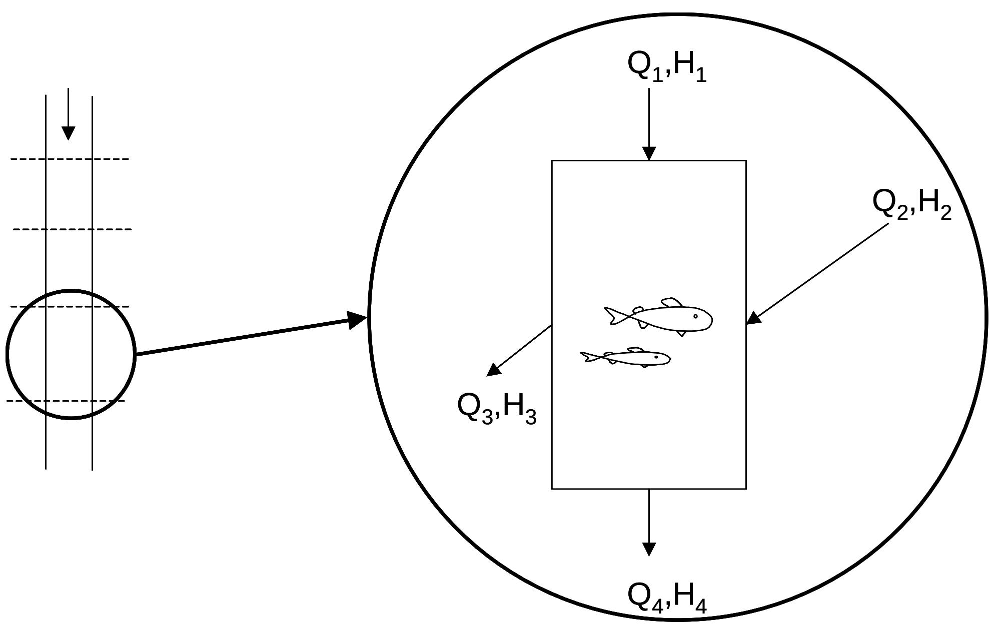


Figure 2. Idealized river system with water and heat inputs Q and H

Instream flow and water temperature are model inputs that can be changed by combinations of different restoration and management decisions, such as environmental water transfers, managing agricultural return flows, and planting shade yielding riparian vegetation to decrease water temperature (Table 3). Water transfers could provide environmental water when and where it is needed, and may improve environmental protection. Changing the time of day that agricultural return flows are added to a river may influence water temperature. Likewise, changing the location of return inflows to a river might add flexibility for managing instream flow and water temperature. Riparian trees and shrubs can shade a river to reduce water temperature in sunny regions where grazing or other land use practices have removed native vegetation. Riparian vegetation is assumed to not affect instream flow significantly.

Table 3. Decision variables affecting instream flow and water temperature

Decision Variables	Effect on Input
Environmental water transfers (WT)	↑ flow
Manage Return Flows (RF)	↓ water temperature?
	Δ flow locally
Riparian Shading (RS)	↓ water temperature

Simple steady state model for 1 fish species

For the first model, the objective is to maximize the population potential of an out-migrating fish species ($F_{a=5}$) over all reaches, r, under steady state hydrology and weather conditions (eq. 1). In the model, fish can move freely between reaches in all age classes, although in this may not be the case for small fish in reality. Fish are limited by age-related mortality and habitat-related carrying capacity for all ages.

$$\text{Max } F = \sum_r F_{a=5,r} \tag{1}$$

subject to:

$$\sum_r F_{a,r} \le f_a\left(\sum_r F_{a-1,r}\right), \forall a \tag{2}$$

$$\sum_r F_{a,r} \le \sum_r f_{q,a,r}(Q_r)\, f_{h,a,r}(H_r), \forall a \tag{3}$$

where $F_{a,r}$ is surviving fish of age a at reach location r, Q_r is instream flow for reach r, H_r is water temperature for reach r, $f_{q,a,r}(Q_r)$ is the carrying capacity factor for age a and reach r as a function of flow, $f_{h,a,r}(H_r)$ is the carrying capacity factor for fish age a and reach r as a function of reach temperature, and f_a is the survivor fraction (1-mortality) of fish age a.

Equation 2 is a fish demography constraint, limiting fish in an age class to survivors of the previous age class. (There can be no more fry than there were eggs, etc.) Equation 3 is a carrying capacity constraint bounding the fish population by instream flow and water temperature conditions. This multiplicative carrying capacity constraint may be too constraining, but is used for illustrative purposes here. Flow is a surrogate for habitat in this example, and more flow does not necessarily equate to better habitat in real systems. In fact, if flow increases, but water temperature remains too high, fish habitat is inadequate (although the constraint used here is less constraining if only flow increases).

If flow and temperature for each reach were subject to direct control, the optimization of fish population potential with respect to Q_r and H_r would be given by the Lagrangian equation:

$$L = \sum_r F_{a=5,r} + \sum_a \lambda_{r,a} \left[(\sum_r F_{a,r} - f_a(\sum_r F_{a-1})) \right] + \sum_a \sum_r \lambda_{a,r} \left[(F_{a,r} - \sum_r f_{q,a,r}(Q_r) f_{h,a,r}(H_r)) \right] \quad \textbf{(4)}$$

Conditions which maximize adult populations would occur when,

$$\frac{\partial L}{\partial Q_r} = 0 = -\sum_a \sum_r (\lambda_{a,r} \frac{f_{h,a,r}(H_r) \partial f_{q,a,r}(Q_r)}{\partial Q_r}), \forall q, h, a, r \quad \textbf{(5)}$$

$$\frac{\partial L}{\partial H_r} = 0 = -\sum_a \sum_r (\lambda_{a,r} \frac{f_{q,a,r}(Q_r) \partial f_{h,a,r}(H_r)}{\partial H_r}), \forall q, h, a, r \quad \textbf{(6)}$$

Solution will return the change in the maximum potential number of surviving out-migrant fish ($F_{a=5}$) for independent changes in water temperature (H_r) or instream flow (Q_r). The two equations for water temperature and flow are used to seek an optimum balance to maximize fish returns. For this example, assume instream flow and water temperature can only be altered by restoration decisions regarding environmental water transfers, managing return flows, or riparian shading. Thus, instream flow and water temperature inputs can be calculated as functions of water and habitat management decisions, using the presumed equations below.

$$H = (H_{WT} WT_r) + (H_{RF} RF_r) + (H_{RS} RS_r) + H_r \ , \forall \mathrm{r} \quad \textbf{(7)}$$

$$Q = (Q_{WT} WT_r) + (Q_{RF} RF_r) + Q_r \ , \forall \mathrm{r} \quad \textbf{(8)}$$

where WT_r are water transfers in or out of reach r; RF_r are return flows to reach r; RS_r is riparian shading in reach r; H_{WT}, H_{RF}, and H_{RS} are the change in water temperature for each unit change in decision variables (WT_r, RF_r, RS_r); and Q_{WT}, and Q_{RF} are the change in flow for each unit change in decision variables (WT_r, RF_r). H_r is additional heating or cooling for reach r, and Q_r is additional inflows or outflows for reach r. Water temperature could also be modeled more explicitly using a mass balance on thermal energy:

$$Q_r T_r = Q_{r-1} T_{r-1} + \sum_i Q_{WT_i} T_{WT_i} + \sum_i Q_{RF_i} T_{RF_i} + \sum_i Q_{RS_i} T_{RS_i} \quad \textbf{(9)}$$

Making linearity assumptions consistent with small changes, the independent effects of changes to H and Q could then be estimated.

$$\Delta H_r = (\Delta H_{WT} WT_r) + (\Delta H_{RS} RS_r) + (\Delta H_{RF} RF_r) \ , \forall \mathrm{r} \quad \textbf{(10)}$$

$$\Delta Q_r = (\Delta Q_{WT} WT_r) + (\Delta Q_{RF} RF_r) \ , \forall \mathrm{r} \quad \textbf{(11)}$$

Equations 7 and 8 can be substituted into the carrying capacity constraint (eq. 3) and optimization of out-migrating fish can be resolved with water transfers (WT_r), return flow management (RF_r), and riparian shading (RS_r) as decision variables.

$$\text{Max } F = \sum_r F_{a=5,r} \tag{12}$$

subject to:

$$\sum_r F_{a,r} \leq f_a(\sum_r F_{a-1,r}), \forall a \tag{13}$$

$$\sum_r F_{a,r} \leq \sum_r f_{q,a,r}\left[((Q_{WT}WT_r) + (Q_{RF}RF_r) + Q_r)\right] f_{h,a,r}((H_{WT}WT_r) + (H_{RF}RF_r) + (H_{RS}RS_r) + H_r), \forall q,h,a \tag{14}$$

Resolving the optimization for the restoration decision variables WT_r, RF_r, and RS_r gives the Lagrangian equation:

$$L = \sum_r F_{a=5,r} + \sum_a \lambda_a \left[(\sum_r F_{a,r} - f_a(\sum_r F_{a-1})) \right] + \sum_a \lambda_a [\sum_r F_{a,r} - \sum_r f_{q,a,r}((Q_{WT}WT_r) + (Q_{RF}RF_r) + Q_r) f_{h,a,r}((H_{WT}WT_r) + (H_{RF}RF_r) + (H_{RS}RS_r) + H_r)] \tag{15}$$

The conditions for maximizing adult fish populations are then:

$$\frac{\partial L}{\partial WT_r} = 0 = -(\lambda_{a,r} \frac{\partial f_{q,a,r}(Q_{WT}WT_r) f_{h,a,r}(H_{WT}WT_r)}{\partial WT_r}) + f_{q,a,r}(Q_{WT}WT_r)\frac{\partial f_{h,a,r}(H_{WT}WT_r)}{\partial WT_r}, \forall q,h,a,r \tag{16}$$

$$\frac{\partial L}{\partial RF_r} = 0 = -(\lambda_{a,r} \frac{\partial f_{q,a,r}(Q_{RF}RF_r) f_{h,a,r}(H_{RF}RF_r)}{\partial RF_r}) + f_{q,a,r}(Q_{RF}RF_r)\frac{\partial f_{h,a,r}(H_{RF}RF_r)}{\partial RF_r}, \forall q,h,a,r \tag{17}$$

$$\frac{\partial L}{\partial RS_r} = 0 = -(\lambda_{a,r} \frac{\partial f_{h,a,r}(H_{RS}RS_r)}{\partial RS_r}), \forall h,a,r \tag{18}$$

Solving for the restoration decision variables (WT_r, RF_r, RS_r) leads to essentially the same solution as solving for instream flow (Q_r) and water temperature (H_r), but this is a more direct method of formulation. This formulation also provides shadow values (the price of an additional unit) for marketed water, managed return flows, or riparian shading. This information is valuable for planning and management because it allows managers to directly compare restoration alternatives in the context of instream flow and thermal advantages of restoration decisions, while providing cost estimates for additional restoration when changes are made to systems. This helps managers understand trade-offs and associated costs between restoration decisions.

Due to the difficulty of valuing environmental goods using a market system, environmental values were not explicitly included in the objective function. Rather restoration cost estimates are represented as a budget constraint. In this way, the cost of improving aquatic conditions for fish can be estimated with a cost for each unit change to a restoration decision variable, where:

$$B = \sum_r (c_{WT}WT_r) + (c_{RS}RS_r) + (c_{RF}RF_r) \tag{19}$$

where B is total budget; c_{WT}, c_{RS}, and c_{RF} are unit costs; WT_r is water transfers; RS_r is riparian shading; and RF_r are return flows for each reach.

Steady state model with 1 immobile age

This model is similar to the previous example, except the first age class cannot move freely between reaches, representing the egg stage of fish life history. Habitat conditions where eggs are laid must be sufficient for them to survive to the next age class. If habitat conditions in a particular reach are poor (elevated temperatures, dessicated redds, etc.), fewer or no eggs in that reach will survive. Fish in subsequent ages can move freely between reaches.

The objective function and fish demography constraints remain the same as the previous model (eq. 1 and 2). A new immobile egg age carrying capacity constraint is added (eq. 20) which changes the carrying capacity constraint for the subsequent age classes from the previous model (eq. 21).

$$F_{r=x,a=1} \leq f_{r=x,a=1}(Q_{r=x}, H_{r=x}) \tag{20}$$

$$\sum_r F_{r,a>1} \leq \sum_r f_{r,a>1}(Q_r, H_r) \tag{21}$$

where r = x specifies a specific reach number, a = 1 is the egg life history stage of fish, and a > 1 are all life history stages after the egg stage.

Seasonal Variation Model

To build upon the previous model, time is added to accommodate fish habitat conditions that can change on the order of the chosen timestep.

$$\text{Max } F = \sum_r F_{a=5,r} \tag{22}$$

subject to:

$$\sum_r F_{t,a,r} \leq f_{r,a}(\sum_r F_{t,a-1,r}), \forall a \tag{23}$$

$$\sum_r F_{t,a,r} \leq \sum_r f_{t,a,r}(Q_{t,r}, H_{t,r}) - harvest, \forall t, r, a \tag{24}$$

where t = time and all other parameters have been previously described. In reality, instream flow and water temperature are highly variable. This model allows for conditions to change (hourly, daily, weekly, monthly, or seasonally depending on the chosen time step) to take into account the variability of instream flow and thermal conditions. Different timesteps target different processes that result in flow and temperature variability. For instance, an hourly timestep captures atmospheric heating of water temperature, a daily model captures heavy rainfall and periods of warm or cold weather, and a seasonal timestep captures fish life history, irrigation use, or seasonal flow and thermal changes. More than one year of flow and water temperature input data is needed to understand how changes in water year type, such as wet or dry years, alter fish habitat.

Simple Steady State Fish Habitat Optimization Models

To evaluate options for managing environmental WUE to improve fish habitat, two connected optimization models were created using Microsoft Excel Solver. These models are similar to the previous examples, but are simpler, and are used primarily to introduce methods and concepts that are further explored in the following chapters of this dissertation. These models illustrate flow and water temperature changes affecting fish productivity by returning agricultural tailwater to different river reaches, or discharging it into an evaporation pond. This provides a proof of concept model showing how managing agricultural tailwater may enhance instream conditions for fish.

Methods

The first model optimizes fish habitat capacity to assess how changes in instream flow and water temperature may theoretically affect habitat conditions for fish. Fish habitat data is passed to the second model, which optimizes out-migrant fish given habitat conditions. Both models use a simplified river reach with one diversion and one spring inflow (Figure 3). For this example, fish habitat exists only in the mainstem reaches, not in the spring, return flows, or irrigation channels. These models do not include a time component, although rate of heating longitudinally is consistent with daily values.

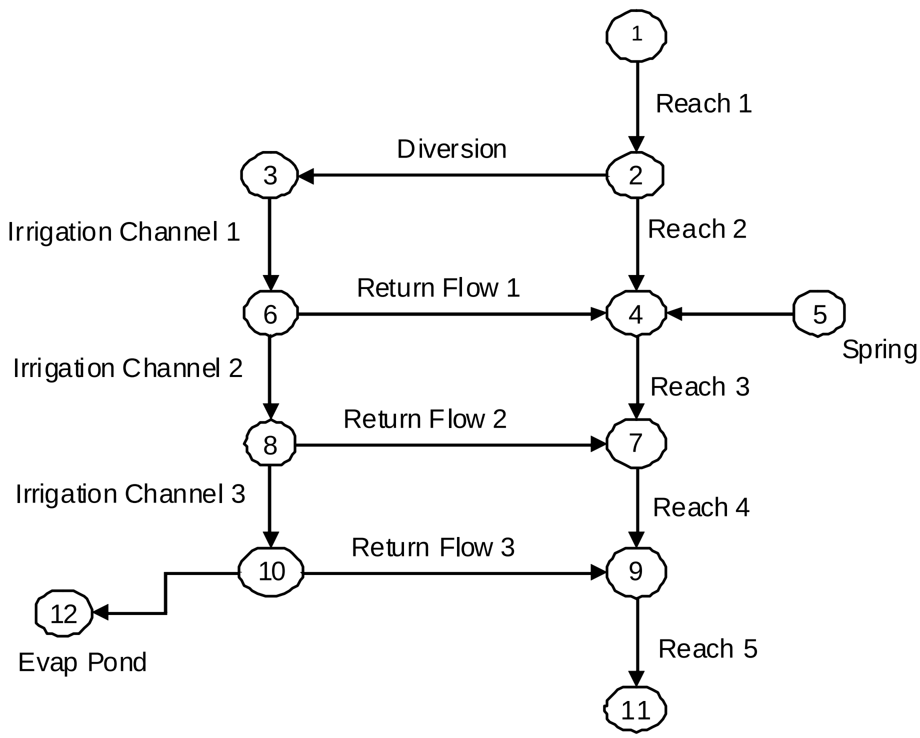


Figure 3. System schematic for habitat capacity model

The models maximize fish survivorship (eq. 25) subject to conservation of mass for instream flow (eq. 26). For water temperature, a mass balance approach is used for conservation of thermal energy (eq. 27); although rate of heating is treated as a pseudo conservative constituent, where the heat budget is simplified by directly placing net daily heating increments on the system (i.e. 0.5ºC per reach). Thus, the heating rate is not affected by changes in flow, although rate of heating can be specified per reach by the user (Table 4). Although the underlying physical processes of the river are not explicitly modeled, the rate of heating in a mainstem river and an irrigation channel are expected to differ due to surface area, depth, travel time, extent of riparian shading, etc. In these models, instream flow and water temperature alone influence fish habitat capacity (eq. 28). All other indicators of habitat quality, such as substrate, predation, competition, disease, food abundance, cover, etc. are ignored.

$$Max\sum_i \sum_j S_{i,j} \qquad \textbf{(25)}$$

subject to:

$$\sum_j Q_{ij} = \sum_i Q_{ij} + b_j \qquad \textbf{(26)}$$

$$\sum_j H_{ij} = \sum_i \frac{Q_{ij} * H_{ij}}{Q_{ij}} + \Delta H_{ij} \qquad \textbf{(27)}$$

$$\sum_j S_{ij} \le \sum_i f_{qij}(Q_{ij}) * f_{hij}(H_{ij}) \qquad \textbf{(28)}$$

where S is fish survivorship, i and j are nodes, Q_{ij} is flow from node i to node j, H_{ij} is water temperature from node i to node j, and b is additional inflow (i.e. springs, return flows).

Table 4. User specified parameters and example values

User Specified Parameters	Location	Example Values
Flow	Reach 1	70
	Spring	10
	Diversion	20
Water Temperature	Reach 1	16
	Spring	15
Rate of Heating	Reach 2 - 5	0.5ºC / reach
	Irrigation Channel 1 -3	1ºC / reach
Maximum Fish/Reach	Reach 1 - 5	500 / reach
Number of In-Migrating Fish	Reach 1 - 5	75 / reach

Additional user specified inputs include the flow and water temperature boundary conditions at reach 1 and the spring, the diversion quantity (water temperature in the diversion is the same as reach 2 water temperature), the rate of heating through the mainstem river and the irrigation channel (Table 4). It is assumed no heating occurs on the return flow links. After initial inflow and water temperature are specified by the user, the model routes water based on mass balance of flow and thermal energy and the user specified rate of heating. Logistic regression then predicts fish survival from the instream conditions of each reach. The maximum number of fish per reach is specified by the user, but is 500 fish per reach for all model runs discussed below. Likewise, the number of in-migrating fish is assumed to be 75 fish per reach for all model runs.

For this example, a three-dimensional logistic surface was developed to relate fish survivorship to instream flow and water temperature using ideal water temperature values for coho salmon (Moyle, 2002; CDFG, 2002; CBSED, 2005). As temperature increases and flow decreases, fish survivorship decreases (Figure 4). The logistic surface can be represented as a continuous logistic surface using equation 29. Under real-world conditions, the habitat capacity surface in Figure 4 would be bell-shaped on both the flow and water temperature axes, because cold water temperatures reduce fish productivity and high flow conditions scour redds or increase velocity, reducing habitat quality. However, for this example, the surface was simplified.

$$S = C * \left(\frac{\left(\frac{k^n}{k^n + T^n} \right)}{1 + e^{a - rQ}} \right) \qquad \textbf{(29)}$$

where C is a constant, and k, n, a, and r, are model parameters with values of 100, 17, 14, 5.3, and 0.1 respectively.

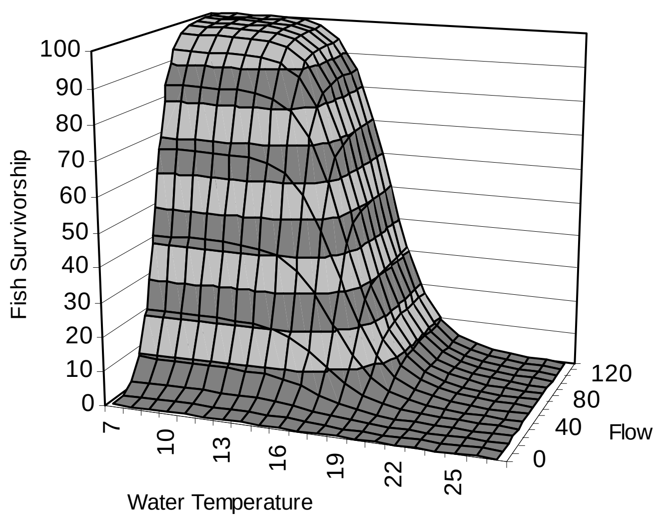


Figure 4. Fish survivorship logistic surface

Fish survivorship can also be represented as a matrix of discrete values to define the percentage of fish survival as a function of instream flow and water temperature (Figure 5). The habitat capacity model optimizes the return flow path that water should take for the highest possible fish survivorship. In general maximizing instream flow and minimizing temperature in all river reaches maximizes fish survivorship.

Flow, cfs	Water Temperature, C: 7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	1	1	1	1	1	1	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0
20	4	4	4	4	4	4	3	3	3	2	2	1	1	0	0	0	0	0	0	0	0
30	9	9	9	9	9	9	9	9	8	6	5	3	2	1	0	0	0	0	0	0	0
40	21	21	21	21	21	21	21	20	18	15	11	7	4	2	1	1	0	0	0	0	0
50	43	43	43	43	42	42	42	40	36	30	21	13	7	4	2	1	1	0	0	0	0
60	67	67	67	67	67	66	65	63	57	47	33	21	12	6	3	2	1	1	0	0	0
70	85	85	85	85	84	84	83	79	72	59	42	26	15	8	4	2	1	1	0	0	0
80	94	94	94	94	93	93	92	88	80	66	47	29	16	9	5	2	1	1	0	0	0
90	98	98	98	98	97	97	95	92	83	68	49	30	17	9	5	3	1	1	0	0	0
100	99	99	99	99	99	98	97	93	84	69	50	31	17	9	5	3	1	1	0	0	0
110	100	100	100	100	99	99	97	93	85	70	50	31	17	9	5	3	1	1	0	0	0
120	100	100	100	100	100	99	98	94	85	70	50	31	17	9	5	3	1	1	0	0	0
130	100	100	100	100	100	99	98	94	85	70	50	31	17	9	5	3	1	1	0	0	0
140	100	100	100	100	100	99	98	94	85	70	50	31	17	9	5	3	1	1	0	0	0

Figure 5. Fish survivorship matrix based on instream flow and water temperature conditions

Limitations

Optimization modeling is useful because it is a simple method to relate instream flow, water temperature, and fish habitat. Any changes to the system, in this case alternative flow paths or use of evaporation ponds, can quickly be compared for improvements to fish habitat. However, this approach has limitations. Travel time and rate of heating are not explicitly modeled, rather the input data for heating rate by reach can be varied by the user. Also for this proof of concept model, the length of river reaches is not defined. Some of these limitations are resolved in Chapter 5, which discusses numerical modeling of the Shasta River, and explicitly models physical processes that alter water temperature through space and time. The model discussed here does not portray an actual river system, but is used to show that optimization modeling is a helpful tool to evaluate environmental water use, and how alternative management of traditional water uses can enhance environmental conditions.

Microsoft Excel Solver sometimes had difficulty optimizing habitat conditions. This non-linear formulation had many decision variables (all potential flow and water temperature values for each link). As a result, iteration in Microsoft Excel Solver would often stop at a good solution, but perhaps not the best, and it was necessary for the user to ensure that results were optimal.

Modeling Sets

Two modeling sets test the theoretical habitat capacity model. In modeling set 1, river reaches heated 0.5°C per reach and irrigation channels heated 1°C per reach. This heating mimics solar radiation on partially shaded river reaches and irrigation channels with little or no vegetation. Total fish habitat and the optimal flow path were analyzed for flows between 10 cfs -120 cfs and water temperatures between 7 - 24°C. In modeling set 2, solar heating conditions were reversed; river reaches heated 1°C per reach, and irrigation channels heated 0.5°C per link. The second case represents river reaches in full sun and partially shaded irrigation channels. This possibility is likely where irrigation channels are narrow and low lying vegetation (reeds, grasses, cattail, etc.) shade channels. In modeling set 2, total fish survivorship and optimal flow path were analyzed for flows between 10 cfs

– 150 cfs and water temperatures between 7°C -26°C (to see if results changed with more widely varying inputs).

Spring inflow and diversion quantities were identical in the two model runs, although diversion quantity changed in both runs based on boundary inflow. When boundary inflow was 10 cfs, 20 cfs, and 30 cfs, diversions were 2 cfs, 10 cfs, and 15 cfs respectively for both modeling sets. Diversions remained at 20 cfs when initial inflow was greater than 40 cfs. A spring always contributed 10 cfs of 15°C water.

Habitat Capacity Model Results

Fish habitat was slightly improved when river reaches heated more slowly than irrigation channels, as occurred in modeling set 1. In modeling set 1 (Figure 6a), the area under the 20%, 40%, 60%, 80%, and 100% fish survival isoquant lines is greater than modeling set 2 (Figure 6b). As stated above, in both runs a 10 cfs, 15°C spring entered the mainstem river at reach 3. This constant cool water influx improved fish habitat when flows were low and water temperatures were high. These results imply that in these instances, it would be better to focus on riparian shading for the mainstem river rather than irrigation channels.

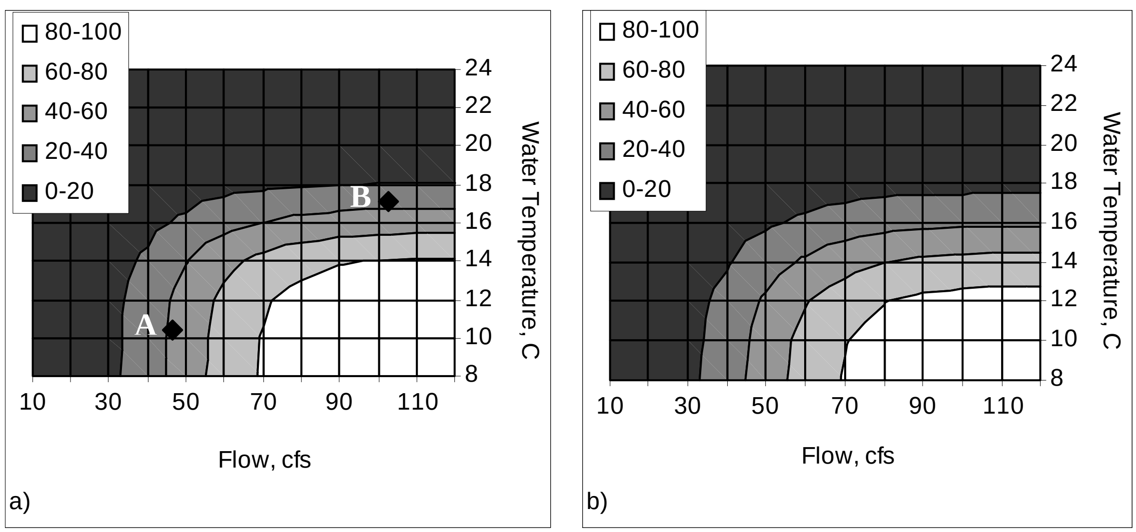


Figure 6. Fish habitat isoquants (by percentage) for a) modeling set 1; and b) modeling set 2

Figure 6 illustrates efficient production decisions for improving fish habitat in this theoretical example. In general, fish habitat conditions are improved by moving from top left to bottom right. The slope of isoquants determines how to most efficiently improve instream habitat. For example, at point A, where isoquant slope is steep, fish survival would most easily be improved by increasing instream flow through acquiring environmental water transfers or not-too-warm return flow. Where isoquants have a relatively flat slope, such as point B, cooling water temperature through additional riparian shading or return flow management is the most effective way to improve habitat conditions and increase fish survival. Here increasing flow is not useful for fish.

Trade-offs can be made to sustain fish habitat under variable conditions. To reach 80% fish survivorship for modeling set 1, upstream boundary condition inflow of at least 70 cfs is needed if upstream boundary condition water temperature is less than 11°C. However, if water temperature in reach 1 is 14°C, at least 100 cfs is needed to maintain habitat conditions. To attain 80% fish survival with rapid heating from solar radiation in mainstem river reaches, as occurred in modeling set 2, initial inflow must be 70 cfs if water temperature in reach 1 is 9°C. If initial water temperature nears 13°C, flow must increase to 100 cfs in modeling set 2 to maintain 80% fish survival (for this model, additional instream flow is assumed to improve fish habitat, which is not necessarily accurate for real-world river systems).

Modeling sets also differed in the optimal path of water through the system. When the mainstem river heated more slowly than irrigation channels, the first return flow channel (return flow into reach 3) was always optimal if initial water temperature was 12°C or less (Figure 7). As initial water temperature increased when instream flow was high, habitat conditions were best maintained by routing warm return flows into an evaporation pond rather than returning them to the river. When initial water temperature was high and instream flow was low, water temperature had to reach 18°C before removing return flows to an evaporation pond became optimal. Although this is a theoretical example, this illustrates the dynamic nature facing water management decision making and implies it is not always beneficial to increase instream flow, and evaporation ponds can sometimes benefit instream habitat.

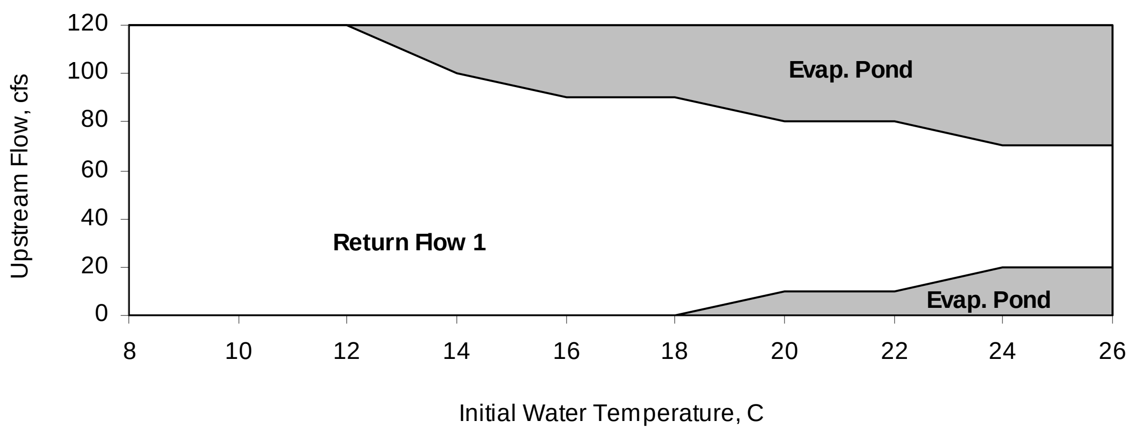


Figure 7. Optimal return flow path with variable initial flow and water temperature conditions for modeling set 1

The best path for return flow water was more variable when mainstem river reaches heated more quickly than irrigation channels for modeling set 2 (Figure 8). When initial inflow was less than 80 cfs, returning agricultural flows at the first channel was best unless water temperature was very high. At higher flows, fish habitat was best if the second return flow channel was used. As fish habitat conditions deteriorated with very high water temperature or very low instream flows, dual solutions were common. In these instances, initial conditions were so poor that any way return flows were routed made little difference. Once some level of minimum flows is achieved, thermal management becomes more important.

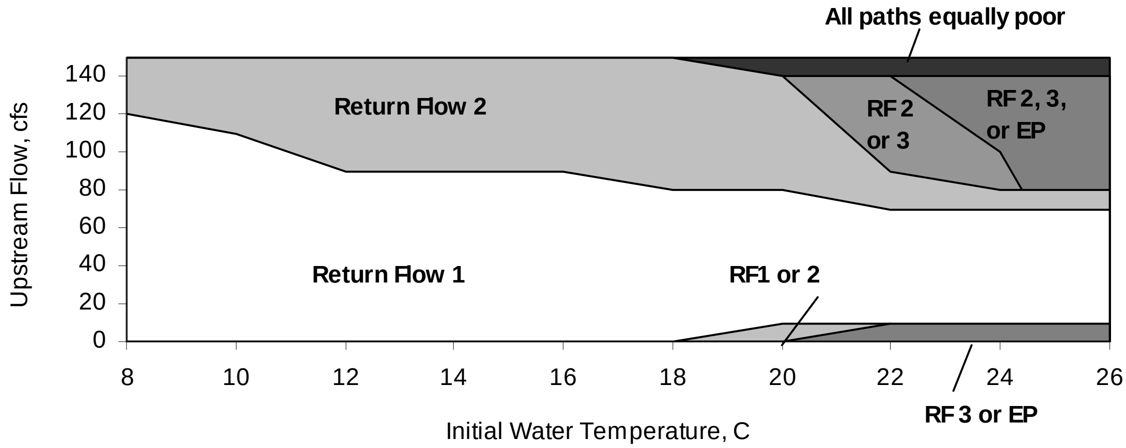


Figure 8. Optimal return flow path with variable initial flow and water temperature conditions for modeling set 2

Fish Model Conclusions

After fish habitat is optimized for a given set of conditions in the habitat capacity model, fish survivorship data is passed to the fish model. The fish model maximizes total out-migrating fish from all river reaches (eq. 1). It is constrained by fish demography (there could be no more fish in an age group than existed in the previous age class) (eq. 2), and total reach capacity based on the instream flow and temperature conditions from the habitat capacity model (eq. 3).

The first age class of fish (spawners) is constrained to specific reaches by the user, meaning spawners must stay at redds and cannot move between reaches. Subsequent age classes can move freely between reaches (and movement does not take up reach capacity). This model uses the same schematic as the habitat capacity model (Figure 3). It is assumed fish only live in the river reaches (not in irrigation channels or return flows). Maximum fish per reach is user specified, but is set to 500 fish per reach in all model runs (meaning each reach could have 100 out-migrating fish if the same reach had 100 spawners, 100 eggs, 100 fry, and 100 juveniles). Fish mortality occurs only from poor instream habitat conditions. The expected fish mortality between age classes is not modeled, and there is an unrealistic 1:1 relationship between all age classes (1 spawner, has 1 egg, produces 1 fry, 1 juvenile, and 1 out-migrant smolt). Given perfect habitat conditions 75 spawners would produce 75 out-migrating fish.

The fish model is sensitive to user specified fish values, particularly for the number of spawners specified in each reach. Too many spawners waste reach capacity, decreasing space for later age classes including out-migrating fish. Since there are 5 age classes of fish, fish production is most efficient if spawners use 1/5 of total capacity. More than 1/5 production of fish in any one age class simply reduces capacity for later reaches. To counteract this problem in the model, the number of spawners per reach (which is user specified) was always less than 1/5 of total reach capacity. In this illustrative model, different age classes consume the same carrying capacity; although in reality the life stages of most fish species do not occur simultaneously.

Results in the fish model could be improved if a time component were added, so that different age cohorts use river reaches at different times. This would make habitat capacity more straightforward to interpret. Results would also be more interesting if instream flow and water temperature requirements were variable among different age classes. This would help tie the fish model with the habitat capacity model so that conditions in different reaches might benefit or hinder some age classes, creating bottlenecks in reaches or age classes. As a final improvement, age class 1 could be changed to in-migrating fish and not assigned to reaches by the user. Then age class 2 could be the immobile egg stage, possibly making the model less sensitive to user defined initial conditions.

Optimizing River Systems by Use

Another strategy for improving environmental WUE might be to specialize rivers or tributaries for environmental or economic water uses, where some river systems are highly developed for traditional water resources and others are left undeveloped for habitat. This deviates from traditional carrying capacity theory, which states people and resource use should be dispersed to not surpass natural thresholds where ecosystems are dramatically changed (Catton, 1986). Water resources in California have been developed using this theory, evidenced by the fact that the Smith River is the only major undammed river in the state (USFS, 2006). However, maintaining viable ecosystems everywhere while simultaneously operating for water distribution, water treatment, recreation, and hydropower is proving costly in terms of money, water, and species (Lackey, 1999). Specializing some river systems for environmental protection and others for development may be more environmentally and economically effective.

This idea is not new. Graf (2001) advocates this idea by simply stating restoration should begin with systems that are the easiest to restore. Mar (1981, 1998) specifically recommends continuing development in already degraded systems instead of moving on to more pristine areas, arguing concentration of resource use is preferable to dispersion. He warns against slow degradation of all systems, believing a more conservative strategy is to protect some waters and severely degrade others (Mar, 1998).

The following model illustrates total benefit from fish production and water supply over two rivers. This simple example has two rivers, one human water demand region (with water deliveries from both rivers), and potential fish habitat downstream from the water demand region on both rivers (Figure 9). It is possible to use each river for both water supply and fish production, or specialize the rivers where the focus is on one of the demands.

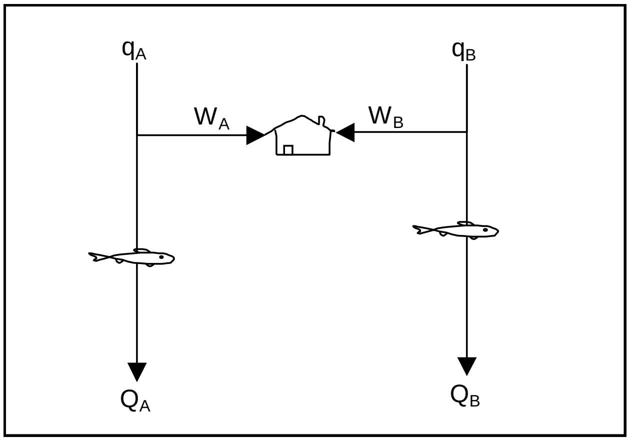


Figure 9. Water supply and fish production on two rivers

Using the simplified system in Figure 9, the objective function is to maximize total benefit from fish production and water supply withdrawal.

$$MaxZ = F_A(Q_A) + F_B(Q_B) + \alpha E(W_A + W_B) \quad \textbf{(30)}$$

subject to

$$q_A = W_A + Q_A \quad \textbf{(31)}$$

$$q_B = W_B + Q_B \quad \textbf{(32)}$$

$$W_A, W_B, Q_A, Q_B \geq 0 \quad \textbf{(33)}$$

$$W_T = W_A + W_B \quad \textbf{(34)}$$

where Z is total benefit, F is fish, E is water supply, α is a weight between 0 and 1, subscripts A and B are rivers, q is inflow, W is withdrawal for water supply, and Q is instream flow for fish production.

Equations 31-33 are conservation of mass constraints binding instream flow and water supply withdrawals for each river by available inflow, and eliminating the possibility of negative instream flow or withdrawals. Equation 34 specifies that total withdrawals are the sum of individual withdrawals from rivers A and B.

Like previous modeling examples, this problem can also be solved using Lagrange multipliers, assuming unit changes to water supply and fish production are independent.

$$\begin{aligned} L = {} & F_A(Q_A) + F_B(Q_B) + E(W_A + W_B) - \lambda_A(q_A - Q_A - W_A) - \lambda_B(q_B - Q_B - W_B) \\ & - \lambda_{W_A} W_A - \lambda_{W_B} W_B - \lambda_{Q_A} Q_A - \lambda_{Q_B} Q_B \end{aligned} \quad \textbf{(35)}$$

$$\frac{\partial L}{\partial Q_A} = 0 = \frac{\partial F_A(Q_A)}{\partial Q_A} + \lambda_A - \lambda_{Q_A} \quad \textbf{(36)}$$

$$\frac{\partial L}{\partial Q_B} = 0 = \frac{\partial F_B(Q_B)}{\partial Q_B} + \lambda_B - \lambda_{Q_B} \quad \textbf{(37)}$$

$$\frac{\partial L}{\partial W_A} = 0 = \frac{\partial E}{\partial W_A} - \lambda_A - \lambda_{W_A} \qquad \textbf{(38)}$$

$$\frac{\partial L}{\partial W_B} = 0 = \frac{\partial E}{\partial W_B} - \lambda_B - \lambda_{W_B} \qquad \textbf{(39)}$$

This solution maximizes total benefit from these rivers by changes to instream flow and water supply withdrawals from rivers A and B. If equations 36 or 37 bind, then this system is limited by instream flow for fish production on river A or B, respectively. If equations 38 or 39 bind, then this system is limited by withdrawals for water supply from rivers A or B, respectively.

It is optimal to concentrate all instream flow for fish production into one river and all water supply withdrawal from the other river when all the following conditions are met:

- a change in fish production at any location with respect to instream flow at that location is positive ($\frac{\partial F_i}{\partial Q_i} > 0$),
- larger instream flows benefit fish production relatively more than small instream flows ($\frac{\partial^2 F_i}{\partial Q_i^2} > 0$),
- a change in water supply from a change in total water withdrawal is negligible when total withdrawal is greater than inflows on river A ($\frac{\partial E}{\partial W_T} \approx 0$ for $W_T > q_A$),
- a change in water supply with respect to a unit change in total withdrawal is greater than the change in fish production from a unit change in instream flow when total water withdrawals are less than inflows on river A ($\frac{\partial E}{\partial W_T} > \frac{\partial F_i}{\partial Q_i}$ for $W_T \le q_A$).

Graphically, where fish and economic production are convex functions with water allocation, it is sometimes optimal to concentrate all fish flows into one river and all water supply withdrawals from another river (Figure 10). Under these conditions, allocation of 50% of each river to economic production, and 50% of each river to fish production does not change economic productivity, but dramatically reduces fish production (Figure 11).

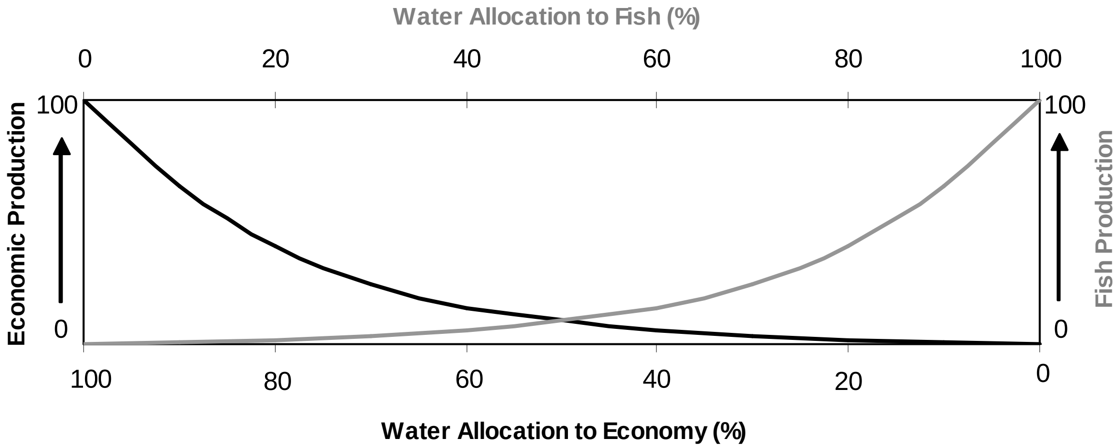


Figure 10. Efficient stream specialization for fish and economic production (100% of river 1 to economic production, and 100% of river 2 to fish production)

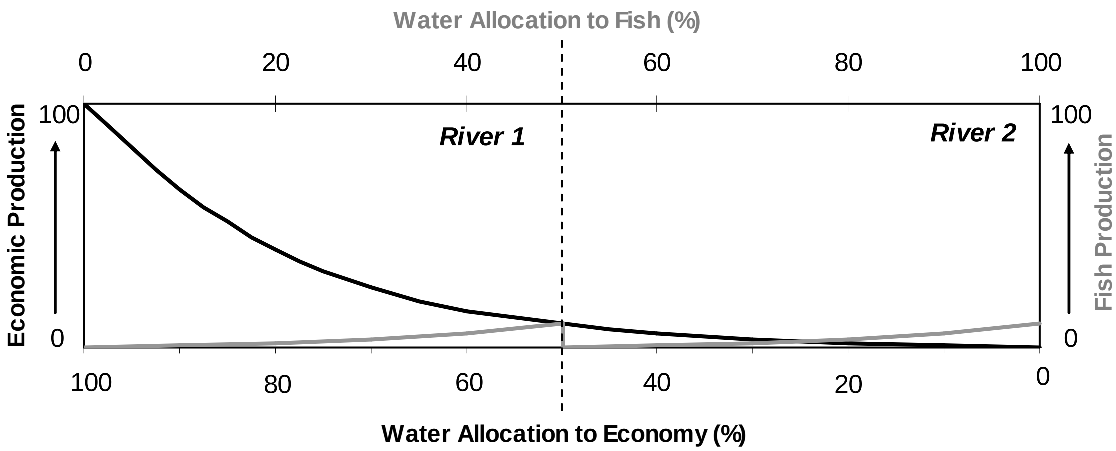


Figure 11. Inefficient fish and economic production (50% from each river to economic and fish production)

When economic production increases quickly with the initial units of water allocation and there is little benefit from additional water, it is no longer efficient to specialize rivers for economic and environmental water uses (Figure 12). In this case both economic and fish production do well with approximately 40% of each stream going toward economic water uses.

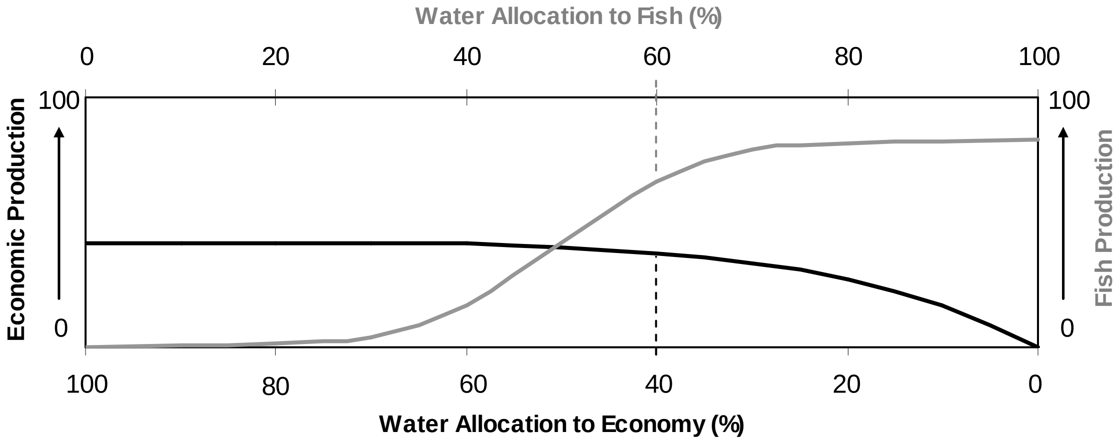


Figure 12. Efficient with 40% of each stream going toward economic uses

Environmental WUE Limitations

Although still a new idea, environmental WUE has strong critics. In part, this stems from fear that environmental WUE strategies will be used to reduce environmental water allocations. For this reason, it is more appropriate to think of environmental WUE as a management strategy than a conservation strategy (Begley et al., 2006). Skeptics are even reluctant to quantify environmental benefits and values, perhaps for fear of this forming a basis for reducing their rights, as occurred with American Indian water right holders (Checchio and Colby, 1993). Although many methods exist to use market approaches to quantify environmental goods, it is notoriously difficult to do well. For this reason, direct economic valuation of environmental goods was not used, rather restoration cost estimates are compared to aid environmental protection decision-making and tradeoffs.

The methods discussed in this chapter call for a more proactive approach to environmental restoration and management. This fundamentally differs from the 'crisis management' approaches inherent in environmental legislation, such as the Endangered Species Act. Proactive management requires long-term restoration commitments, instead of opportunistic restoration. More importantly, this also necessitates direct management of the environment. This is problematic when we do not fully understand ecosystem processes and functions. Yet it is important to emphasize that humans are already managing aquatic environments, typically after species are threatened or important habitats are in danger of disappearing. A proactive approach to restoration could possibly improve ecosystem health and save species, but may also introduce liability if managed environments function poorly. Adaptive management may provide a framework to increase understanding of natural systems while limiting liability (Holling, 1978).

Specializing rivers for environmental or traditional water uses may be difficult politically and economically where property values differ along waters with various specified uses, or where different American Indian tribal water and fishing rights occur on rivers or reaches. In these instances, specializing rivers may amount to choosing between tribes, communities, or constituents (Mar, 1981; Cornwall, 2005). Specializing rivers also may be inappropriate when endemic species or unique habitats exist in different rivers or tributaries because these rivers cannot be considered equal substitutes. In these instances, restoration efficiency must be kept in mind. Unsuccessful restoration projects for endangered species or critical habitat on many rivers might sometimes be worse for the environment than a small number of successful projects.

Finally, river systems are far more complex than the simplified models presented here. In the first scenario, instream flow and water temperature are the only variable inputs under steady state conditions. Simple rates of heating do not translate to different thermal mass and travel time under variable flow rates. Averaged water temperature does not take into account diurnal variation that makes water temperature lethal to fish during warm summer months. All biotic interactions and other habitat requirements are ignored. The second scenario also uses a very simplified system, with only a single urban water demand region. In reality, numerous and competing uses for water make problems and decisions infinitely more complex. Nevertheless, these models should suffice to illustrate the potential value of environmental WUE in theory.

Conclusions

Further understanding of how to best manage natural river systems when water is scarce is badly needed for California and much of the American West. Additional water for the environment is unfortunately both difficult to secure and increasingly costly. Hoping for more water for environmental protection is noble, but may not be effective for improving the diversity, function, and quality of natural river systems. This chapter includes discussion of environmental WUE and considers a framework to enhance environmental conditions when new water allocations are unlikely. The interdisciplinary approach taken here combines methods from the geosciences, biology, engineering, and economics to analyze and develop new strategies for managing competing environmental and human water use.

Despite water allocations, extensive (and expensive) restoration programs, and good intentions, fish and other species are disappearing throughout California and the U.S. Established methods of environmental protection and enhancement do not always achieve desired goals. Potential restoration alternatives must be evaluated for the environmental benefit that can be realized, and held accountable for the resources used with each alternative. In this way, informed decisions regarding optimal restoration strategies can be made. Quantifying the benefits of proposed restoration activities is difficult, but when possible it must be considered to evaluate the most effective restoration alternatives and conserve precious water and dollar resources.

Optimization modeling provides a useful tool to test the potential of environmental WUE because many potential solutions can be evaluated quickly, and innovative management ideas can be incorporated easily. Optimization allows water managers to estimate trade-offs between decision variables, helping to determine the most water efficient and cost effective environmental protection. This contributes to increased understanding of environmental WUE management decisions, planning strategies, and successful implementation of aquatic restoration programs.

California and other western states face water shortages and difficult allocation decisions. In the future, as new water supplies become difficult to secure, attention will focus on how to best manage existing supplies for multiple traditional and environmental uses. As urban and agricultural water use sectors have had to adapt to use water supplies more efficiently, so too must the environmental sector. Managing environmental water use to improve efficiency will become essential, and innovative management practices will likely become more widespread.

References

AFRP (Anadromous Fish Restoration Program). 2001. Final Restoration Plan for the Anadromous Fish Restoration Program. Prepared by USFWS and AFRP under CVPIA for DOI. Available at http://www.delta.dfg.ca.gov/afrp/documents/finalrestplan.pdf. Accessed 5/06.

Anderson, J. 3/22/2006. "The Eternal Dustbowl: Paying for the sins of L.A.'s water barons has created a half billion dollar boondoggle". L.A. Weekly.

Bay Delta Authority website. 2006. Available at http://calwater.ca.gov/Programs/EcosystemRestoration/Ecosystem_BattleCreek.asp. Accessed 6/06.

Begley, L., K. Edwards, S. Heard, L. Walling. 2006. Managed Environmental Water Use Efficiency: Evidence in Practice. Prepared for CDWR. U.C. Berkeley Goldman School of Public Policy. Berkeley, CA.

Brown, R.A., G.B. Pasternack. 2008. Engineered channel controls limiting spawning habitat rehabilitation success on regulated gravel-bed rivers. Geomorphology 97(2008): 631 – 654.

Burke, S.M., Adams, R.M., Wallender, W.W. 2004. "Water banks and environmental water demands: Case of the Klamath Project". Water Resources Research. 40(W09S02).

CBSED (City and Borough of Sitka Electric Department). 2005. Draft Habitat Suitability Curves for Coho Salmon and Steelhead Trout, Sawmill Creek Alaska. Prepared for Blue Lake Hydroelectric Project, FERC No. 2230, Relicensing.

CDFG (CA. Dept. of Fish and Game). 2002. Status Review of California Coho Salmon North of San Francisco. Report to the California Fish and Game Commission. Available online: http://www.dfg.ca.gov/fish/Resources/Coho/SAL_CohoStatusRpt.asp. Accessed 6/2008.

CDWR (CA. Dept. of Water Resources). 2005. California Water Plan Update, Volume IV. http:www.waterplan.water.ca.gov/docs/cwpu2005/Vol_4/04-Environment/V4PRD2-mewue.pdf. Accessed 5/06.

Cardwell, H., J.I. Jager, M.J. Sale. 1996. Designing Instream Flows to Satisfy Fish and Human Water Needs. Journal of Water Res. Planning and Mgmt: 122(5), 356-363.

Catton, W. 1986. Carrying capacity and the Limits to Freedom. Paper prepared for Social Ecology Session 1, Xl World Congress of Sociology. New Delhi, India.

Checchio, E., B.G. Colby. 1993. Indian Water Rights: Negotiating the Future. Water Resources Research Center, The University of Arizona College of Agriculture. Tucson, Arizona.

Cohon, J.L. 1978. <u>Multiobjective Programming and Planning</u>. Academic Press. San Francisco, CA. 333 pp.

Collins, A. R. Rosenberger, J. Fletcher. 2005. The economic value of stream restoration. Water Resources Research (41).

Cornwall, W. 11/7/2005. "Is salmon money flowing right way?" The Olympian.

Davenport, D.C., P.E. Martin, R.M. Hagan. 1982. Evapotranspiration from riparian vegetation: Conserving water by reducing saltcedar transpiration. Journal of Soil and Water Conservation.

Deas, M.L. 2000. Application of Numerical Water Quality Models in Ecological Assessment. Doctoral Dissertation, University of California, Davis. Davis, CA.

Deason, J., Fast, J., Schroeer, L., Turner, B., van Staveren, R. 2004. Considering Water Use Efficiency for the Environmental Sector. Prepared for CDWR, Statewide Water Planning Branch. UC Berkeley, CA.

Disco, C. 2002. Remaking "Nature": The Ecological Turn in Dutch Water Management. Science, Technology, & Human Values, 27(2): 206 – 235.

Environmental Defense. 1999. Mitigation Banking as an Endangered Species Conservation Tool. Available at http://www.defenders.org/habitat/highways/new/trb/EDreport.pdf. Accessed 6/06.

Fell, A. 2000. Settlement Reached in Long-running Putah Creek Water Dispute. U.C. Davis News and Information. May 24, 2000. Available online: http://www-news.ucdavis.edu/search/news_detail.lasso?id=5115. Accessed 8/2008.

Fleckenstein, J., E. Suzuki, G.E. Fogg. 2001. Options for conjuctive water management to restore fall flows in the Cosumnes River basin, California. Integrated water resources management: 272, 175-182.

Gillilan, D.M., T.C. Brown. 1997. Instream Flow Protection: Seeking a Balance in Western Water Use. Island Press, Washington, D.C. and Covelo, CA.

Graf, W.L. 2001. *Dam*age Control: Restoring the Physical Integrity of America's Rivers. Annals of the Assoc. of American Geographers. Blackwell Publishers. Madden, MA.

GCMRC (Grand Canyon Monitoring and Research Center). 1999. Draft Science Plan to Accompany Environmental Assessment for a Temperature Control Device for Glen Canyon Dam. Prepared for U.S.B.R. Available at http://www.usbr.gov/uc/rm/amp/tcd/pdfs/tcdplan10.pdf. Accessed 5/06.

Harden, B. 3/7/2004. "Salmon's Return Spurs Debate on Spill at Dams, Need for Costly Measure Questioned". Washington Post.

Hart, J. 1996. Storm Over Mono: The Mono Lake Battle and the California Water Future. University of California Press. Los Angeles and Berkeley, CA.

Heitmeyer, M.E., D.P. Connelly, R.L. Pederson. 1989. The Central, Imperial, and Coachella Valleys of California. Pgs 475 – 505 in L.M. Smith, R.L. Pederson, R.H. Kaminski (eds.). Habitat Management for Migrating and Wintering Waterfowl in North America. Texas Tech University Press. Lubbock, TX.

Hillier, Y., G. Lieberman. 1967. Introduction to Operations Research. Holden-Day Publishers. San Francisco, CA. 639 pp.

Holling, C.S. (ed.). 1978. Adaptive Environmental Assessment and Management. The Blackburn Press. Caldwell, NJ.

Hollinshead, S. 2005. Optimization of Environmental Water Account Purchases with Uncertainty. Master's Thesis, University of California, Davis. Davis, CA.

Jones & Stokes. 2003. "Environmental Water Program Conceptual Proposal Guidance Document". Prepared for the Environmental Water Program. Sacramento, CA.

Kauffman, J.B., R.L. Beschta, N. Otting, D. Lytjen. 1997. An Ecological Perspective of Riparian and Stream Restoration in the Western United States. Fisheries: 22(5), 12-24.

Kelley, Jr., J.R., M.K. Laubhan, F.A. Reid. 1993. Options for Water-level Control in Developed Wetlands. U.S. Dept. of the Interior National Biological Survey, Fish and Wildlife Leaflet 13.4.8: Waterfowl Management Handbook. Washington, D.C.

Kier Associates. 1999. Battle Creek Salmon and Steelhead Restoration Plan. Prepared for the Battle Creek Working Group. Sausalito, CA.

Labadie, J.W. 2004. Optimal Operation of Multireservoir Systems: State-of-the-Art Review. Journal of Water Resources Planning and Management 130(2): 93-111.

Lackey, R.T. 1999. "Salmon Policy: science, society, restoration, and reality". Environmental Science and Policy. 2 (99): 369-379.

Lankford, B.A. 2003. "Environmental Water Requirements: A Demand Management Perspective". Journal of the Chartered Institution of Water and Environmental Management. 17(1): 19-22.

Lowney, C.L. 2000. River Temperature Dynamics in the Sacramento River: Measurement and Modeling. Doctoral Dissertation, U.C. Davis. Davis, CA.

Mar, B.W. 1981. Dead Is Dead – An Alternative Strategy for Urban Water Management. Urban Ecology: 5 (1980/1981) 103-112.

Mar, B.W. 1998. System Requirements for Water Resource Systems.

Moyle, P.B. 2002. Inland Fishes of California. University of California Press. Berkeley and Los Angeles, California.

Nehlsen, W., Williams, J.E., Lichatowich, J.A. 1991. "Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington". Fisheries. 16(2):4-21.

Null, S.E., J.R. Lund. 2006. Reassembling Hetch Hetchy: Water Supply Without O'Shaughnessy Dam. Journal of the American Water Res. Assoc. 42(2):395-408.

Null, S.E. 2007. Water Use by Yellow Starthistle. Unpublished data.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, J.C. Stromberg. 1997. The Natural Flow Regime: A paradigm for river conservation and restoration. Bioscience: 47(11), 769-784.

Richter, B.D., J.V. Baumgartner, J. Powell, D.P. Braun. 1996. A Method for Assessing Hydrologic Alteration within Ecosystems. Conservation Biology: 10(4), 1163-1174.

Shuford, W.D., G.W. Page, J.E. Kjelmyr. 1998. Patterns and Dynamics of Shorebird Use of California's Central Valley. The Condor: 100(2), 227-244.

Sommer, T., B. Harrel, M. Nobriga, R. Brown, P. Moyle, W. Kimmerer, L. Schemel. 2001. California's Yolo Bypass: Evidence that flood control can be compatible with fisheries, wetlands, wildlife, and agriculture. Fisheries: 26, 6-16.

Sommer, T.R., W.C. Harrell, M.L. Nobriga and R. Kurth. 2003. Floodplain as habitat for native fish: Lessons from California's Yolo Bypass. Pages 81-87 *in* P.M. Faber, editor. California riparian systems: Processes and floodplain management, ecology, and restoration. 2001 Riparian Habitat and Floodplains Conference Proceedings, Riparian Habitat Joint Venture, Sacramento, California.

Sunding, D.L. 2003. Making Room for the Environment in California Water Policy: Ten Years of the Central Valley Project Improvement Act. University of California Giannini Foundation of Agricultural Economics. 7(2):9-11.

USBR website. Available at http://www.usbr.gov/mp/nepa/nepa_base.cfm?location=ncao. Accessed 5/06.

US EPA. 1995. Federal Guidance for the Establishment, Use and Operation of Mitigation Banks. Federal Register: 60(228):58605-58614

US EPA. 2006. Iron Mountain Mine – Success through planning, partnerships, and perseverance. Washington DC.

USFS (U.S. Forest Service) North Coast Rivers website. 2006. Available at http://www.fs.fed.us/r5/publications/water_resources/html/north_coast_rivers.html. Accessed 6/06.

Vermeyen, T.B. 1997. Modifying Reservoir Release Temperatures Using Temperature Control Curtains. Proceedings of Theme D: Energy and Water: Sustainable Development, 27th IAHR Congress, San Francisco, CA, August 10-15, 1997. Available at http://www.usbr.gov/pmts/hydraulics_lab/tvermeyen. Accessed 5/06.

Williams, J.C., C.S. ReVelle, D.J. Bain. 2003. A decision model for selecting protected habitat areas within migratory flyways. Socio-Economic Planning Sciences: 37(2003): 239-268.

Winternitz, L., J. White. 2001. "Implementing the CALFED Bay-Delta Program's Environmental Water Account – California".

Yolo Basin Foundation website. 2006. Available at http://www.yolobasin.org/wildlife.cfm. Accessed 6/06.

Zavaleta, E. 2000. The Economic Value of Controlling an Invasive Shrub. Ambio: 29(8): 462-467.

Chapter 3: Background – California's Shasta River

This chapter introduces California's Shasta River, describing the physical setting and historic conditions that made the Shasta River a productive salmon fishery, and discusses current conditions that have led to the decline of native salmon and steelhead. The current status of salmonid fish species is described, and previous studies relating to the Shasta River are summarized.

The Shasta River, located in California's Siskiyou County, is a tributary to the Klamath River. It is the last major tributary before Iron Gate Dam, the first impoundment on the Klamath River (Figure 13). Since migratory fish no longer have access to the Klamath River above Iron Gate Dam, tributaries such as the Shasta, Scott, Salmon, and Trinity Rivers, are of greater importance to the health and survival of migratory salmonids. Historically, the Shasta River was dominated by numerous cold-water springs, providing ideal, year-round, cool water habitat for salmon and steelhead. Today, surface water diversions, groundwater pumping, and construction of Dwinnell Dam have greatly decreased instream flow and fundamentally altered the hydrograph, while low flow conditions, grazing of riparian vegetation, tailwater returns, and diversion of cool springfed sources have substantially increased dry-season water temperatures (NRC, 2004).

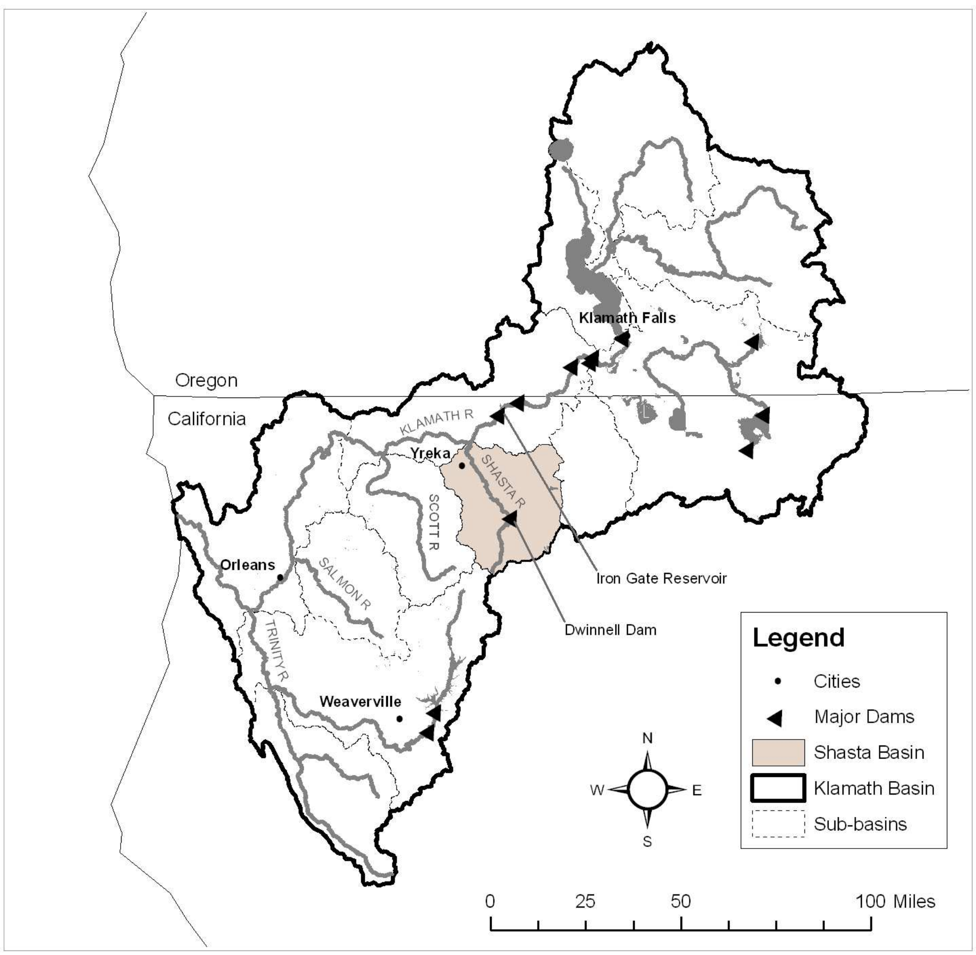


Figure 13. Klamath River watershed, with major dams and tributaries

The Shasta River watershed is approximately 800 mi^2, and the river flows northward from its headwaters to the confluence with the Klamath River (Figure 14). It is bounded by the Salmon and Marble Mountains, Mt. Eddy, and Mt. Shasta. Its headwaters are approximately 70 miles above the confluence with the Klamath River, and include Dale, Eddy, Boles, Beaughton, Carrick, and the upper reaches of Parks Creek. The profile of the Shasta River is steep at its headwaters, followed by a large alluvial valley, and then a steep canyon reach before it joins the Klamath River (Figure 15). The alluvial valley has unique hilly topography caused by a volcanic debris flow from Mt Shasta.

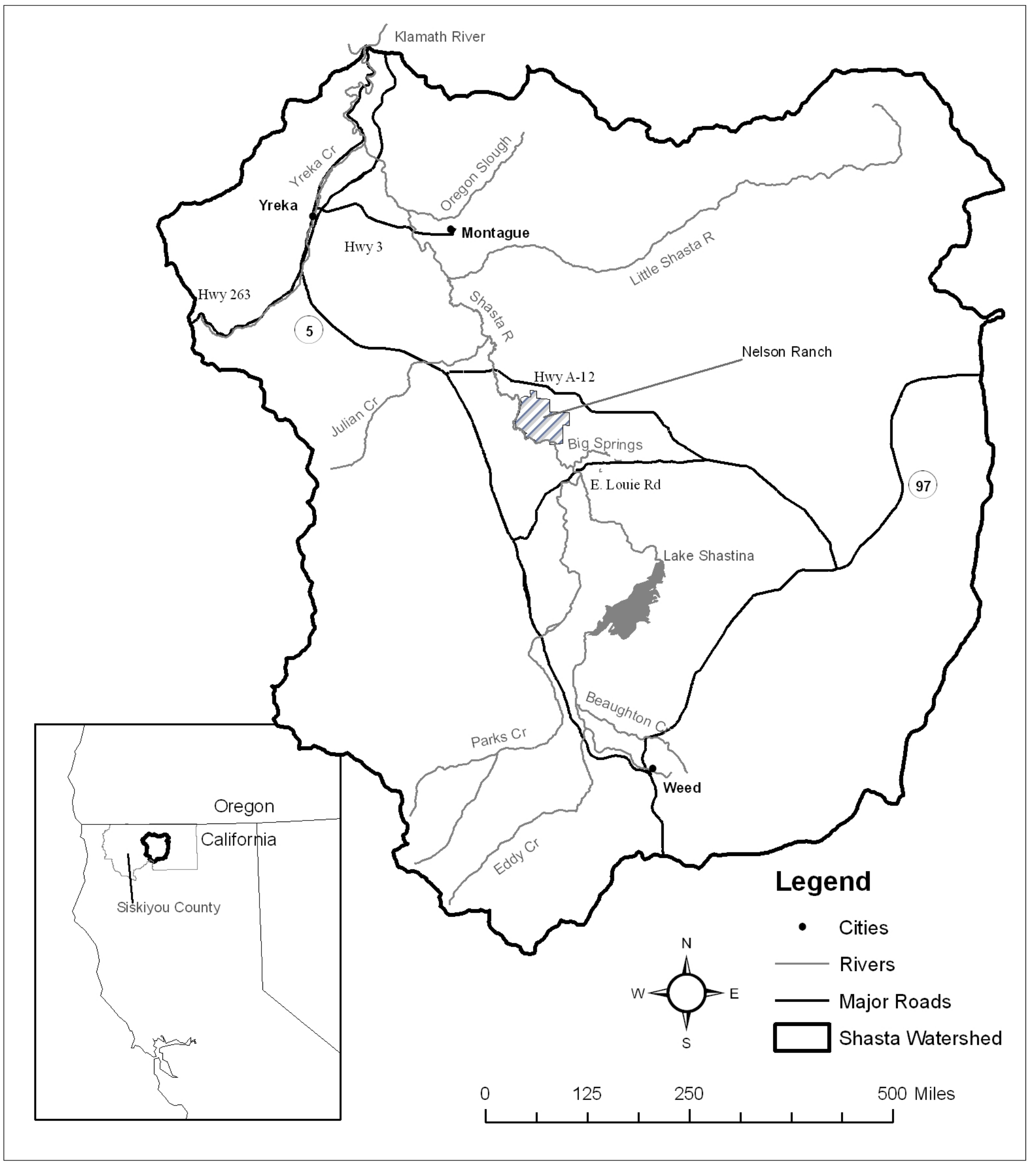


Figure 14. Shasta watershed

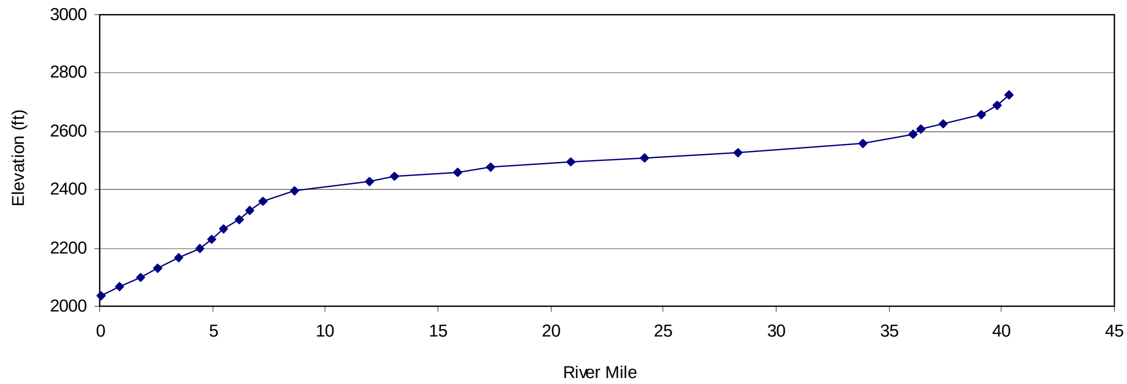


Figure 15. Shasta River longitudinal profile (from Abbott, 2002)

The Shasta River is in the rain shadow of the Salmon and Marble Mountains, making it markedly drier than the neighboring Scott, Salmon, and Trinity watersheds. Precipitation averages 10-18 in/yr in the Shasta Valley, mostly in the form of winter rain and snowfall. Current mean annual unimpaired runoff is approximately 136,000 af. This contribution to the Klamath River is usually insignificant, it accounts for less than 5% of annual runoff at Seiad, and less than 4% of summer runoff at Seiad (USGS, 2008).

An extensive spring system made the Shasta River arguably the most productive salmon and steelhead river in California (Snyder, 1931; NRC, 2004). Prior to water development in the Shasta Valley, the river maintained a year round baseflow of approximately 200 cfs (NRC, 2004). Spring water sources typically are 11-12°C; thus, the spring-fed river provided cool summer water temperatures and relatively warmer winter temperatures, ideal for salmonids (NRC, 2004).

Current Conditions

Dwinnell Dam impounds Lake Shastina, the only major dam on the Shasta River. It is owned and operated by the Montague Water Conservation District (MWCD) to store winter flows, with water rights of 60,000 af, although maximum operating capacity is 50,000 af (Booher et al., 1960s). By most standards, both the dam and the irrigation system are highly inefficient. There is more seepage than there is water delivered to downstream irrigators (NRC, 2004). Such losses may boost groundwater recharge, but may also increase groundwater temperatures somewhat. Reports exist that new springs appeared after filling the reservoir and that springs below Grenada are reduced when reservoir capacity is below approximately 20,000 af (Crabill, pers.comm., 2007; Scott, pers.comm., 2007). The Shasta River immediately below Dwinnell Dam maintains 4-5 cfs from reservoir seepage and springs, with instream flow increasing to 15 cfs three miles downstream of the dam (Scott, pers.comm., 2007). Construction of Dwinnell Dam ended upstream passage for migratory fish at the damsite, reduced geomorphically important peak flows associated with local storms, and reduced gravel recruitment below the dam.

Below Dwinnell Dam, the Shasta River has four major tributaries: Parks Creek, the Big Springs complex, the Little Shasta River, and Yreka Creek (Figure 14). Parks Creek enters the Shasta River approximately seven miles downstream of Dwinnell Dam. MWCD has a 15,000 af water right from Parks Creek, diverting water from Parks Creek into Dwinnell Dam.

The Shasta River, like most California rivers, has annual low flow in early summer through early fall in response to the Mediterranean climate that typifies the region. However, local spring inflows modify this typical seasonal hydrograph below the Big Springs complex. The Big Springs complex is a natural group of springs approximately seven downstream of Dwinnell Dam. Prior to water development, the springs contributed a constant 103 cfs of cool water to the Shasta River (Mack, 1960). Today contributions from the Big Springs complex are approximately 70 cfs (NCRWQCB, 2006). At spring sources, approximately two miles east of the Shasta River, year round water temperature is approximately 12°C; although poor tailwater management can raise water temperature to 25°C at the confluence with the Shasta River.

The Little Shasta River is highly developed with many diversions. Both the Little Shasta River and Yreka Creek contribute minimal inflow to the Shasta River. Small tributaries include Willow and Julian Creeks, and the Oregon Slough. Small creek channels typically become dry during summer months, and Oregon Slough is primarily agricultural return flow. Thus, measured streamflow downstream of Dwinnell Dam is driven by inflow from tributaries (i.e. Parks Creek), discrete natural springs (i.e. Big Springs), and diffuse groundwater.

Land use in the Shasta Valley is primarily grazing and low-value agriculture although urbanization is increasing near Yreka and Montague; and Weed and Lake Shastina are experiencing increasing development pressure (NRC, 2004). Most agricultural land in the Shasta Valley is dedicated to beef production, including dry and irrigated pasture, alfalfa, and some grain production (TNC, 2003). Water use is primarily for agriculture and grazing, but also includes some urban, industrial, recreational, and wildlife uses (Deas et al., 2004). The irrigation season is from April to October, when flow in the river drops from an average of 200 cfs to as low as 20 cfs (USGS, 2008).

There are four major diversions from the Shasta River, belonging to Montague Water Conservation District (MWCD), Big Springs Irrigation District (BSID), Grenada Irrigation District (GID), and the Shasta Water Users Association (SWUA) (Figure 16). MWCD diverts water straight from Dwinnell Dam (Lake Shastina) into the MWCD canal. BSID pumps groundwater upslope of the Big Springs complex. In the 1980s, BSID began pumping groundwater instead of diverting surface water, which contributed to the Big Springs channel subsequently becoming dry (NRC, 2004). GID is located at river mile 30.58 and is the most junior water right holder on the Shasta River. Between April to October, GID diverts approximately 20-42 cfs, depending on the number of pumps operating and water availability. The SWUA diversion is located downstream of the Little Shasta River. From April to October, SWUA typically diverts 42 cfs.

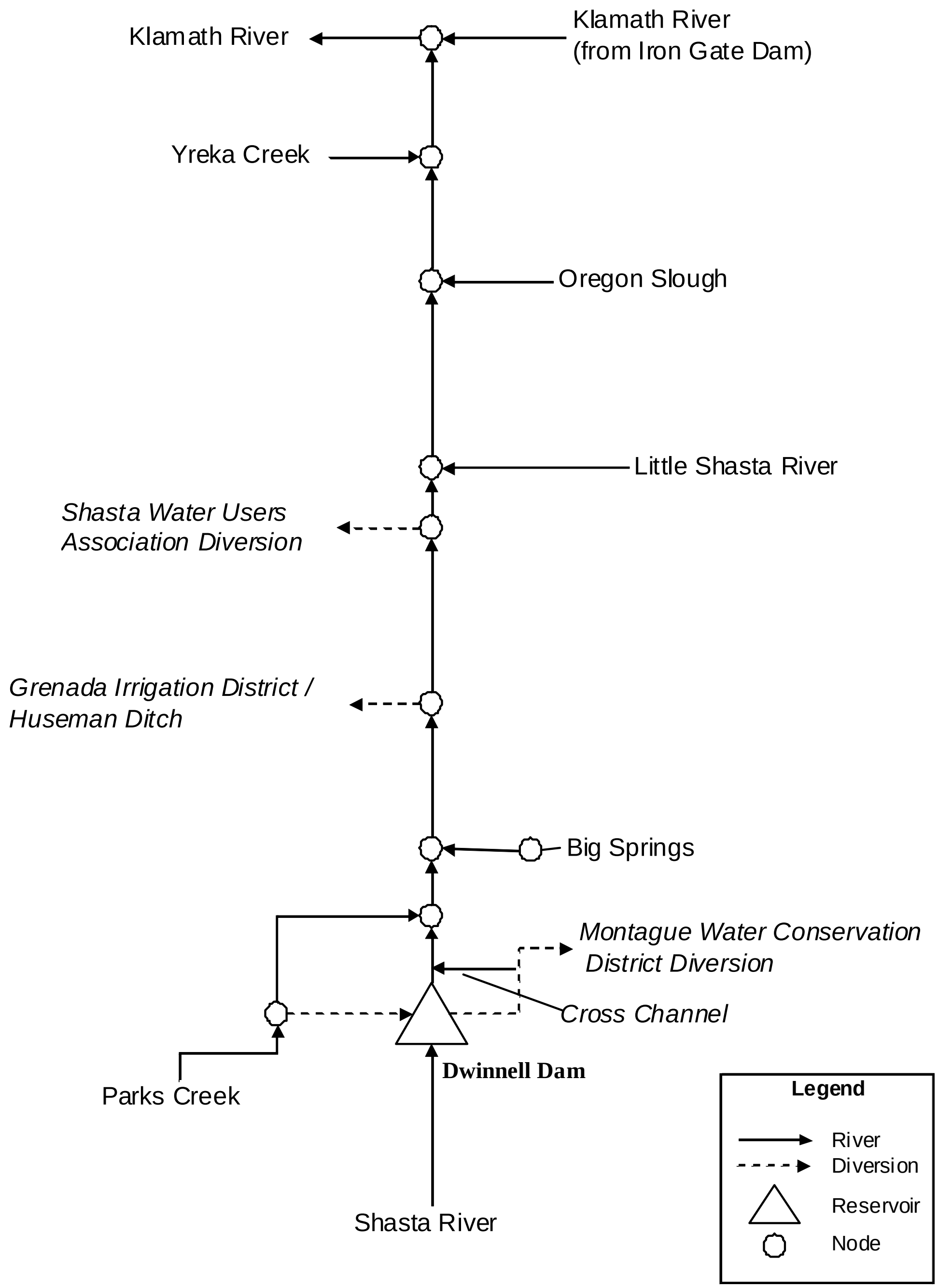


Figure 16. Schematic of major Shasta River inflow and outflow

Numerous small and moderate diversions occur over the length of the Shasta River by individual landowners (Figure 17). According to water rights, maximum allowable diversions are approximately 112 cfs to landowners in the upper Shasta River above Big Springs, 178 cfs in the lower Shasta River, and 92 cfs to landowners along the Little Shasta River (DWR Watermaster report, 2006); however, due to timing and the priority of water rights, less water is typically diverted. The Shasta River has been largely adjudicated since

1934, although riparian water right owners are entitled to additional water not under Watermaster service (DWR Watermaster Report, 2006), and groundwater pumping has not been adjudicated. Since the 1970's, the number of groundwater wells has been increasing in the Shasta Valley (NRC, 2004). Groundwater has not been well quantified in the Shasta Valley and in general is not well understood at this time.

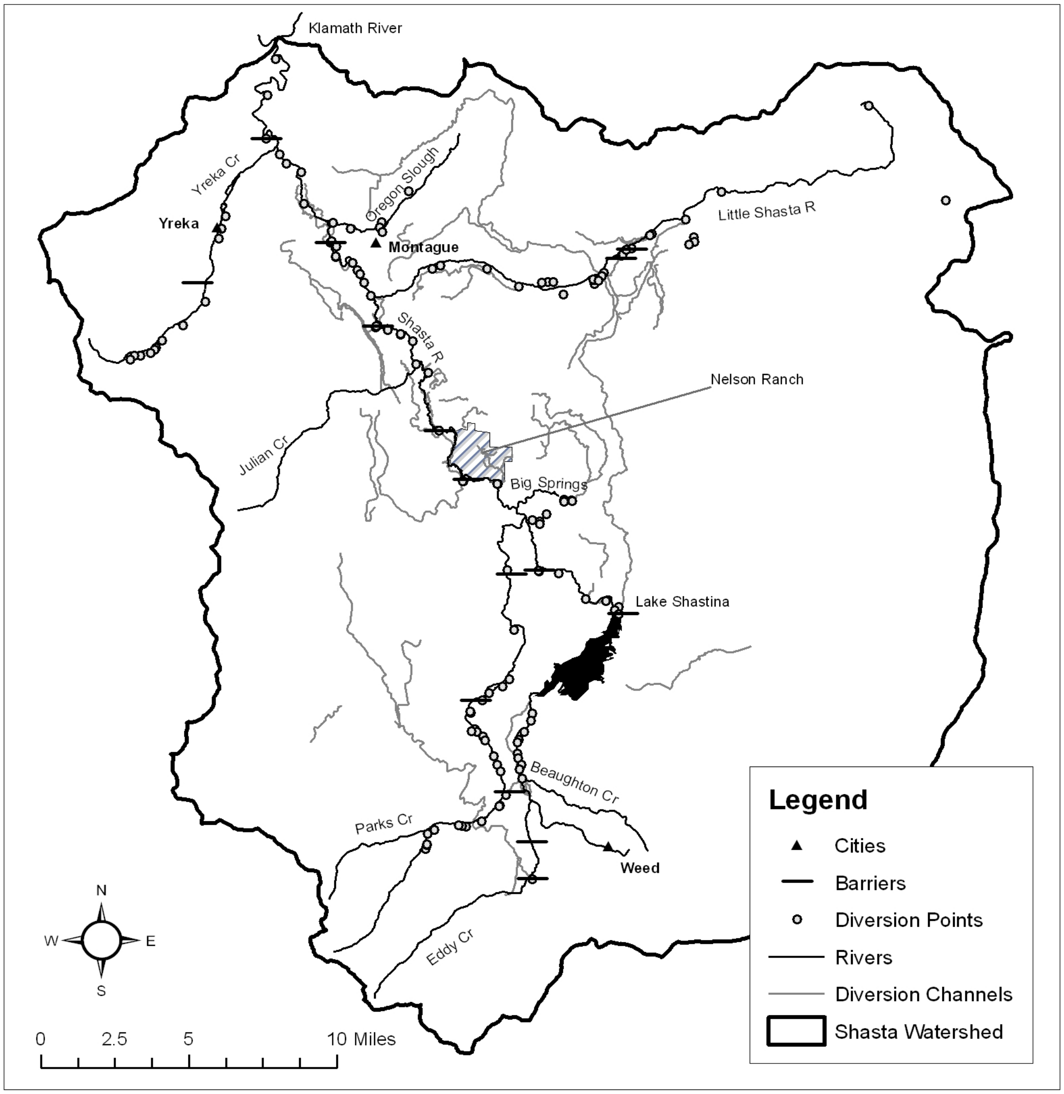


Figure 17. Shasta River barriers and diversion points

The thermal regime of the river has been severely affected by reduced instream flow, diversion of springfed water sources, loss of riparian vegetation, and tailwater return flow. While no records of historic water temperature exist, it is known that optimal water temperature for juvenile salmonids, which were abundant, range from 12 - 18°C (Moyle, 2002). In 2006 – 2007, mean annual water temperature was 17.1°C, and maximum water temperature was 24.5°C at Nelson Ranch. Further downstream weekly average water temperature can exceed 25°C, well above the lethal limit for salmon (Deas et al., 2004).

Temperature conditions in the Shasta River are largely driven by hydrology (and geohydrology) and meteorology. Unique attributes of the system are the temperature signals from substantial spring inflows, which may enter the river either notably warmer, nearly the same, or considerably cooler than ambient water temperature depending on the time of year. The springs create unique thermal conditions when compared to streams without springs.

In general, groundwater-dominated river systems, like the Shasta River, have a more stable flow and thermal regime than those not dominated by groundwater (Sear et al., 1999). Groundwater dominated systems can moderate the influence of meteorological conditions by direct dilution of stable inflow temperatures, as well as increasing the volume of the receiving water. The result is less seasonal variability (Caissie, 2006). Big Springs Creek contributes the majority of spring-derived water, although smaller springs occur upstream of the Big Springs complex. Thus, Shasta River water temperatures and flows are relatively stable in the reach immediately below Big Springs; however, meteorological conditions exert an increasing influence as distance from the Big Springs source increases.

Meteorological conditions are a primary factor driving thermal conditions in the Shasta River, and are exacerbated by low flow conditions. Low flow conditions, prevalent during the summer irrigation season, increase water temperature because a shallow river has less thermal mass and a longer travel time to the mouth. Atmospheric heating then becomes a dominant influence on water temperature. Water temperature response to solar radiation varies seasonally with maximum loading occurring during late spring and summer months when day length is long, solar altitude is at an annual maximum, and cloudy days are few. Air temperature reflects a similar response to seasonal solar radiation (Figure 18). Average summer (6/1 – 9/30) air temperature at Nelson Ranch was 19.4°C during 2007, although daily average air temperature can exceed 25°C (77°F) in July and August. Maximum air temperature was 39.7°C (103°F) on July 10, 2007, and minimum air temperature was –13.5°C (7.7°F) on January 13, 2007.

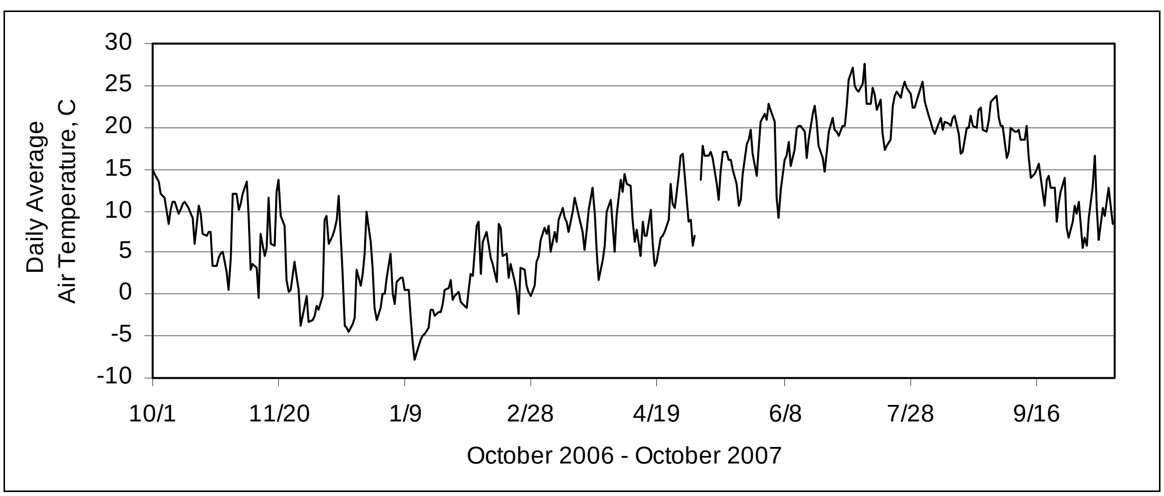


Figure 18. Nelson Ranch daily air temperature

39

Thermal heating is compounded by the current lack of shading by riparian vegetation. Riparian vegetation primarily reduces thermal variability by absorbing and filtering solar radiation, which can provide 95% of the heat input to a river at midday during the summer (Brown, 1970). To a lesser extent, riparian vegetation also moderates water temperature by reflecting back-radiation, reducing wind speed, and altering the air temperature and relative humidity directly above the water surface. Thus, a healthy riparian corridor can maintain cool water temperature by reducing solar transmittance, and slightly increase nightly minimum temperature by absorbing back-radiation.

The Shasta River is a narrow river; so dense riparian vegetation can block or filter solar radiation for much of the river during most of the day. However, grazing along river banks is widespread throughout the Shasta Valley. Riparian vegetation surveys conducted in 1996 (Deas et al.) show that most of the Shasta River had less than two trees every 30m, and sections with no trees were common. Since grazing and agriculture have occurred in the Shasta Valley since the 1800s, the natural vegetative state surrounding the Shasta River is not well understood. However, a full gallery forest along the length of the Shasta River was unlikely, due to anoxic soils throughout the valley. According to riparian vegetation field surveys conducted in 2001 by Abbott (2002), bulrush was found throughout the system where the river was protected from grazing, and average height of bulrush was 3m. Thus, it is likely that bulrush would provide some shading in areas where trees cannot survive if riparian fencing were more extensive. The Shasta Valley Coordinated Resources Management and Planning (SVCRMP) group is currently working with landowners to fence the riparian corridor and plant trees in promising reaches to regenerate a riparian corridor (SVCRMP, 2007). In addition to shading the river, riparian vegetation prevents bank erosion, provides channel habitat complexity, and creates cover and pool habitat (NRC, 2004).

Tailwater return is a significant, but unquantified heat source to the Shasta River during summer months. Return flow can be substantial. For example, 18 cfs is allocated to various water right holders below the DWR weir. Typically return flow and other unknown accretions make up approximately 20 cfs at the weir so reservoir water need not be released (Scott, pers.comm., 2007). Tailwater flows to the river as channelized, overland, and subsurface flow; and most likely, contributes to both Shasta River baseflow and groundwater spring sources, although precise tailwater flows have not been studied. During the afternoon in summer months, tailwater return flow is substantially warmer than the mainstem river because it collects in very shallow channels or is spread over fields. Better return flow management, including water re-use, recharge and evaporation ponds, or timing returns for cold water periods (early mornings) may show promise for reducing instream water temperature.

Although low flow conditions and water temperature are the factors most limiting salmon productivity in the Shasta River; other problems exist (CDWR, 2001; NRC, 2004). In addition to Dwinnell Dam, there are seven small flashboard dams that are used for agricultural diversions during the irrigation season (approximately May through September) (Figure 17). Most diversions have fish screens but do not provide adequate fish passage, act as barriers to upstream migration in spring (when fish are avoiding warmer downstream reaches), increase predation, and have poor local water quality conditions. Dwinnell Dam blocks all access to upper reaches of the Shasta River for migratory fish species. Additionally, Dwinnell Dam has significantly altered the hydrograph and geomorphology of the Shasta River. As stated above, peak flows from winter storms no longer occur, except during infrequent reservoir spills (i.e. 1964 and 1997) (Jeffres et al., 2008). This is significant because in addition to the loss of spawning and rearing habitat from construction of the dam, remaining habitat is affected by reduced gravel recruitment and higher volumes of fine sediment, which fills the interstitial spaces between gravel limiting water and oxygen flow (NRC, 2004). Finally, water temperature is inversely related to dissolved oxygen. While there have been few observations of dissolved oxygen below saturation, each occurrence has coincided with high water temperature (NRC, 2004).

The Shasta River currently does not meet federal water quality standards under the Clean Water Act Section 303(d) because of organic enrichment/low dissolved oxygen and high water temperature. Pollution control plans, called Total Maximum Daily Loads (TMDLs) have been created to identify and control low dissolved oxygen and high water temperature conditions in the Shasta River from the mouth to the headwaters, including all tributaries and Lake Shastina. TMDLs are imposed to limit the pollution or stressors that the Shasta River can receive from point sources, non-point sources, and natural background loading (NCRWQCB, 2006).

Fish species and status

Historically, the Shasta River was a healthy salmon and steelhead river. Four fish species, fall and spring run Chinook (*Oncorhynchus tshawytscha*), coho (*O. kisutch*), and steelhead (*O.giardneri*) were present, with spring run Chinook the most abundant. Fish populations prior to the 1930's are estimated to be approximately 30,000-80,000 Chinook/year, 6000 coho/year, and 1000 steelhead/year (Moyle, 2002; CDFG, 1991; CDFG, 1965). Prior to construction of Dwinnell Dam, the Shasta River was probably already partially degraded from irrigation and timber harvesting practices that began in the 1850's. This is evident in a description by Snyder (1931), as a "stream once famous for its trout and salmon". Dwinnell Dam, located approximately 40 miles above the mouth of the Shasta River, was completed in 1928. Construction of Dwinnell Dam blocked access to upstream habitat leading to the extirpation of spring run Chinook in the Shasta River (Moyle, 2002).

Coho salmon in the Shasta River belong to the Southern Oregon-Northern California coasts Ecologically Significant Unit (ESU), which was listed as federally threatened by the National Marine Fisheries Service (NMFS) in 1997 (Moyle, 2002), and endangered by the California Endangered Species Act in 2003 (NRC, 2004). Since the early 1980's, coho typically number less that 100 fish per year in the Shasta River, illustrating their precarious existence in the basin (reported numbers often do not represent the entire run since fish counting facilities are discontinued during high flow conditions) (CDFG, 2003, SSRT, 2003). Coho spawn in late fall and early winter, fry emerge in early spring, and juveniles rear for an entire year before migrating to the ocean (Figure 19).

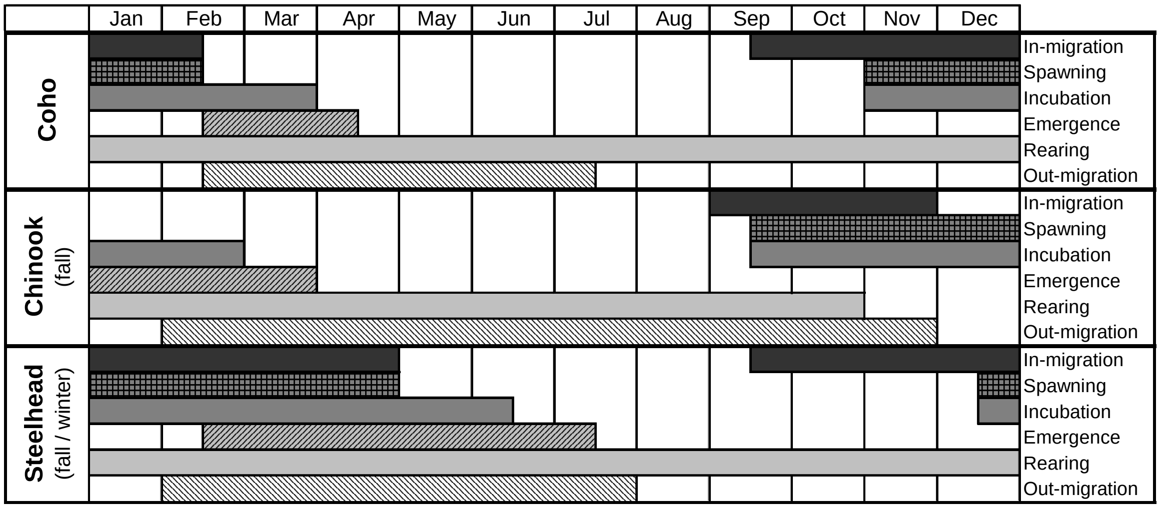


Figure 19. Salmonid life history timing (CDFG, 1997; NCRWQCB, 2006)

Shasta River fall run Chinook belong to the upper Klamath and Trinity Rivers ESU. From 1978 to 2002, Chinook runs have averaged 5,630 returning adults, with a high of 18,731 fish in 1978 and only 553 fish in 1990. The Shasta River fish counting facility normally operates from September to the first week of November, but has been extended to count coho until December (when high flows necessitate removing the facility). Therefore, these numbers accurately estimate the fall run Chinook migration, although it is common to include hatchery fish from Iron Gate or Trinity River Fish Hatcheries (CDFG, 2003). Typically juvenile type I Chinook out-migrate soon after fry emergence, with most migrants leaving by June. However, juvenile Chinook have been observed in the Shasta River throughout summer months, implying type II Chinook (rearing through spring and summer), or type III Chinook (rearing for an entire year before out-migrating) may be present in the Shasta River (CDFG, 1997).

Winter run steelhead from the Klamath Mountains Province ESU remain fairly common in the Shasta River, despite large-scale habitat reduction from the construction of Dwinnell Dam. In 2002, 1,712 juvenile and adult steelhead were observed at the Shasta River fish counting facility (CDFG, 2003).

Major factors related to reduced spawning and the decline of migratory fish species include low flows, increased water temperature, altered river channel geomorphology, periods of low dissolved oxygen, land use changes resulting in a loss of riparian vegetation (and associated cover), barriers to migration such as flashboard dams, and loss of access to habitat above Dwinnell Dam (NRC, 2004; Deas et al., 2004). Coho, fall run Chinook, and steelhead are present during summer and early fall when temperature is limiting (Table 5), although availability of food, cool water refugia, and lack of predators influence survival.

Table 5. Tolerated and lethal water temperature by fish species (Moyle, 2002; Deas et al., 2004)

Species	Upper Tolerance (C)	Lethal (C)
Fall run Chinook	21 -22	23-24
Coho	20	25

Previous and Ongoing Studies

Numerous analyses have assessed factors limiting salmon production in the Shasta River (CDFG, 1997; Deas et al., 2004, NRC, 2004). Some reports gathered existing information to identify information gaps and recommend management practices to restore habitat conditions and salmon populations. Table 6 is a partial listing of recent research and data collection in the Shasta Valley organized by topic. Flow and water temperature studies have been combined into a single heading because flow changes inherently affect water temperature. Preliminary results from only one study are discussed below.

Table 6. Recent Shasta River research, analysis, and monitoring

		Location	Citation
Flow and Temperature		Location	Citation
	Flow and water temperature studies	Basin wide	CDWR, 1964; CDWR, 1985
	Summer and fall flow and temperature gaging	Dwinnell and mouth	Deas et al., 2003a
	Big Spring Complex flow quantification	Big Springs	Deas, 2006
	Simulations of thermal regime from variable flow rates, pulse flows, return flow management, and riparian shading alternatives	4 miles below Dwinnell Dam to mouth	Deas et al., 2003b
Water Quality			
	Shasta River temperature and DO TMDL, including monitoring and modeling studies	Basin wide	NCRWQCB, 2006
	Investigation of water quality conditions in the Shasta River	Dwinnell Dam to the mouth	Gwynne, 1993
	Shasta River water quality study	Basin wide	CDWR, 1986
	Lake Shastina Limnology Study	Lake Shastina	Vignola and Deas, 2005
	Water quality and aquatic habitat characterization	Basin wide	CDWR, 2001
Fish Habitat and Productivity			
	Assessment of fish habitat quality and limitations	Basin wide	CDFG, 1996; Ricker, 1997; USFWS, 1992
	Coho recovery recommendations focusing on agricultural practices and water use	Basin wide	SSRT, 2003
	Shasta River fall Chinook counts (1930-present), partial run sizes also noted for coho and steelhead	Mouth of Shasta River	CDFG, 2003; CDFG, 2002
	Analysis of habitat quality and factors limiting salmon productivity in the Shasta River	Basin wide	CDFG, 1997; Deas et al., 2004; NRC, 2004
Other Data Collection and Monitoring			
	Geometric stream channel characterization (cross section, depth, bankfull width, bank height)	Dwinnell Dam to mouth	Deas et al., 2003a
	Monitoring (river stage and flow, temperature and other water quality, fish snorkel surveys, geomorphology, floodplain and habitat mapping, macroinvertebrate, and aquatic macrophyte productivity)	Nelson Ranch	Jeffres et al., 2008
	Woody riparian vegetation inventory	Dwinnell Dam to mouth	Deas et al., 1996
	Riparian vegetation height and canopy transmittance field sampling	Dwinnell Dam to mouth	Deas et al., 2003a

Historically, limited access to certain river reaches restricted research and monitoring activities. However, in recent years access to several key reaches has been widely available. Since June 2006, extensive research and monitoring has occurred on the California Nature Conservancy's Nelson Ranch. In 2008 numerous other landowners, including landowners on Big Springs Creek, have allowed researchers to monitor the river from their property. This has provided an opportunity to collect data and conduct research over many miles of the Shasta River. UC Davis' Center for Watershed Sciences is conducting an extensive monitoring effort (Jeffres et al., 2008).

According to observations, spawning coho return to either the canyon reach of the Shasta River, approximately four miles above the confluence of with the Klamath River, or the upper Shasta River (Big Springs complex to Dwinnell Dam) (Jeffres et al., 2008). The canyon reach has favorable spawning and rearing habitat conditions during winter, although by summer water temperature and low flow conditions make this reach lethal to coho. Low flows combined with instream barriers caused by flashboard dams make migration upstream into more favorable habitat nearly impossible. This effect may work as an ecological trap, reducing the survival and fitness of coho, with no known environmental cues to warn spawning coho that habitat in the canyon will degrade during summer months (Jeffres et al., 2008).

Discussion

The following chapters examine restoration alternatives to enhance habitat conditions for native fish species. Analysis occurs over multiple spatial scales, with field studies used to understand localized and small-scale conditions, simulation modeling to examine how specific restoration measures affect instream flow and water temperature at different river locations, and optimization modeling to synthesize understanding of restoration alternatives over the entire watershed and compare alternatives based on projected fish habitat.

References

Abbott, A.G.P. 2002. The Effect of Riparian Vegetation on Stream Temperature in the Shasta River. Master's thesis. U.C. Davis. Dept. of Civil and Environmental Engineering.

Booher, L.J., C.E., Houston, A.D. Reed, M.V. Maxwell, S.D. Nelson. 1960s. Analysis of the Montague Water Conservation District. University of California Agricultural Extension Service. Yreka, CA.

Brown, G.W. 1970. Predicting the Effect of Clearcutting on Stream Temperature. Journal of Soils Water Conservation. 25:11-13.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology. **51**, 1389-1406

CDFG (CA Dept. of Fish and Game). 1965. Klamath River Hydrographic Unit. North Coastal Area Investigation, Bulletin 136. Sacramento, CA.

CDFG (CA Dept. of Fish and Game). 1991. Annual report for the Klamath River project. Sacramento, CA. (As cited in Alcorn, 1992).

CDFG (CA Dept. of Fish and Game). 1996. A Biological Needs Assessment for Anadromous Fish in the Shasta River, Siskiyou County, California - DRAFT. Northern Management Area (Area 2), Yreka, CA 96097. June.

CDFG (CA Dept. of Fish and Game). 1997. A biological needs assessment for anadromous fish in the Shasta River, Siskiyou County, California. CA Dept. of Fish and Game, Northern California-North Coast Region. Redding, CA.

CDFG (CA Dept. of Fish and Game). 2002. Status Review of California Coho Salmon North of San Francisco. Report to the California Fish and Game Commission. Available online: http://www.dfg.ca.gov/fish/Resources/Coho/SAL_CohoStatusRpt.asp. Accessed 6/2008.

CDFG (CA Dept. of Fish and Game). 2003. Shasta River Fish Counting Facility, Chinook and Coho Salmon Observations in 2002. Siskiyou County, CA.

CDWR (CA Dept. of Water Resources). 1964. Shasta Valley Investigation. Bulletin No. 87. Sacramento, CA.

CDWR (CA Dept. of Water Resources). 1985. Shasta/Klamath Rivers Water Quality Study.

CDWR (CA Dept. of Water Resources), Northern District. 1986. Shasta/Klamath Rivers Water Quality Study. Sacramento, CA.

CDWR (CA Dept. of Water Resources). 2001. Watershed Study Plan: Water Quality and Aquatic Habitat Characterization in the Shasta River. Prepared by G.L. Boles. Northern District. Red Bluff, CA.

CDWR (CA Dept. of Water Resources) Watermaster Report. 2006. Lower Shasta River, Upper Shasta River, and Little Shasta River service areas. Available online (http://www.nd.water.ca.gov/PPAs/Watermasters/ServiceAreas/ShastaRiver/index.cfm). Accessed 4/2007.

Crabill, B. 4/18/2007. MWCD (Montague Water Conservation District). Personal communication.

Deas, M.L., J. Haas, G.T. Orlob. 1996. Shasta River Woody Riparian Vegetation Inventory. Clean Water Act 205(j) Grant Program, California State Water Resources Control Board and the Shasta Valley Resources Conservation District.

Deas, M.L., A. Abbott, A. Bale. 2003a. Shasta River Flow and Temperature Modeling Project. Prepared for Watercourse Engineering, Inc. Napa, CA.

Deas, M.L., A. Abbott, A. Bale. 2003b. Shasta River Flow and Temperature Modeling Project. Prepared for Watercourse Engineering, Inc. Napa, CA.

Deas, M.L., P.B. Moyle, J. Mount, J.R. Lund, C.L. Lowney, S. Tanaka. 2004. Priority Actions for Restoration of the Shasta River – Technical Report. Prepared for The Nature Conservancy. June 11.

Deas, M.L. 2006. Big Springs Creek and Spring Complex – Estimated Quantification Technical Memorandum. Prepared for the North Coast Regional Water Quality Control Board. Davis, CA.

Gwynne, B.A. 1993. Investigation of the Water Quality Conditions in the Shasta River, Siskiyou County. California RWQCB, North Coast Region. Santa Rosa, CA.

Jeffres, C., E. Buckland, J. Kiernan, A. King, A. Nichols, S. Null, J. Mount, P. Moyle, M. Deas,. 2008. Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California – Water Year 2007. Prepared for the California Nature Conservancy by U.C Davis Center for Watershed Sciences and Watercourse Engineering, Inc.

Mack, S. 1960. Geology and Ground-Water Features of Shasta Valley, Siskiyou County, California. U.S. Geological Survey Water-Supply Paper 1484. Washington, DC: U.S. Government Printing Office.

Moyle, P.B. 2002. Inland Fishes of California. University of California Press. Berkeley and Los Angeles, California.

NCRWQCB (California North Coast Regional Water Quality Control Board). 2006. Staff Report for the Action Plan for the Shasta River Watershed Temperature and Dissolved Oxygen Total Maximum Daily Loads. Santa Rosa, CA. Available online (http://www.swrcb.ca.gov/rwqcb1/programs/tmdl/shasta/060707/staff-report-complete.pdf). Accessed 6/2007.

NRC (National Resources Council). 2004. Endangered and Threatened Fishes in the Klamath River Basin: Causes of Decline and Strategies for Recovery. Committee on Endangered and Threatened Fishes in the Klamath River Basin, National Research Council. The National Academy Press.

Ricker, S.J. 1997. Evaluation of Salmon and Steelhead Spawning Habitat Quality in the Shasta River Basin. CA Dept. of Fish and Game. Inland Fisheries Administrative Report No. 97-9. Arcata, CA.

TNC (The Nature Conservancy). 2003. The Klamath Basin Project Conservation Area Plan.

Scott, J., K. Dick. Shasta Valley CDWR Watermasters. Personal Communication. 4/13/2007.

Sear, D.A., P.D. Armitage, F.H. Dawson. 1999. Groundwater dominated rivers. Hydrological Processes. 13, 255-276.

Snyder, J.O. 1931. Salmon of the Klamath River, California. CA Dept. of Fish and Game Bulletin 34. Sacramento, CA: California State Printing Office. Available at http://www.krisweb.com/biblio/klamath_cdfg_snyder_1931.pdf. Accessed 5/2007.

SVCRMP (Shasta River Coordinated Resources Management and Planning). Shasta River CRMP Home website: http://www.snowcrest.net/shastacrmp/. Accessed 5/2007.

SSRT (Shasta-Scott Coho Salmon Recovery Team). 2003. Shasta and Scott River Pilot Program for Coho Salmon Recovery: with recommendations relating to Agriculture and Agricultural Water Use. Prepared for the CA Dept. of Fish and Game.

USFWS (United States Fish and Wildlife Service). 1992. An Issue Statement on Shasta River Water Use and Possible Impacts on Anadromous Salmonids. Prepared by Doug Alcorn. August.

USGS (US Geological Survey). 2008. Real Time Water Data for California website. Available at http://waterdata.usgs.gov/ca/nwis/rt. Accessed 7/2008.

Vignola, E., M.L. Deas. 2005. Lake Shastina Limnology. Watercourse Engineering, Inc. Prepared for the UC Davis Information Center for the Environment and the North Coast Regional Water Quality Control Board.

Chapter 4: Shasta River Temperature and Flow Monitoring

This chapter summarizes temperature and flow studies undertaken on the Shasta River. Longitudinal temperature monitoring, spatial and temporal thermal diversity studies, tailwater return and spring inflow monitoring were conducted over a five-mile stretch of the Shasta River at the Nature Conservancy's Nelson Ranch. Tailwater was also monitored at Meamber Ranch, near the DWR weir in the lower Shasta River. The longitudinal temperature monitoring was conducted in conjunction with the UC Davis Watershed Sciences Center and Watercourse Engineering, Inc., recording hourly water temperature approximately every 400 meters in the main channel along Nelson Ranch where the Shasta River is well mixed. The spatial and temporal variability studies consist of thermal diversity sampling during summer months to locate and assess local cool water habitat, and lateral transect monitoring during winter and spring months to increase understanding of thermal diversity. Preliminary agricultural return flow and temperature monitoring was conducting at Nelson Ranch and Meamber Ranch, by monitoring the volume and temperature of tailwater return. Flow and water temperature were also recorded at Dream Spring on Nelson Ranch to increase understanding of spring quality and potential in restoration activities. These studies are described in this chapter, with discussion of major findings.

Existing water temperature data for the Shasta River is sporadic, and mainly exists for summer months. However, additional temperature monitoring is currently being conducted at various sites along the length of the Shasta River. Much of the fieldwork undertaken here focuses on filling existing information gaps by exploring longitudinal and lateral thermal variability over the course of a year. Monitoring the thermal conditions and variability of the Shasta River helps increase understanding of thermal conditions on a scale more detailed than can be simulated with computer models. Results from the field studies discussed here will be used to interpret model results, incorporating small-scale temperature diversity and variability.

Elevated summer water temperatures reduce cold-water fish habitat and are known to limit fish survival (NRC, 2004). However, it is important to increase understanding of spatial and temporal thermal variability into winter and spring when native salmon are migrating into the Shasta River to spawn, emerging from redds, rearing, and out-migrating from the Shasta River (Figure 19). Yearlong temperature data is used to create a baseline assessment of thermal conditions in the Shasta River and to understand seasonal changes that may influence restoration activities. The research discussed here contributes to and interfaces directly with monitoring conducted by the UC Davis Center for Watershed Sciences and Watercourse Engineering, Inc. to evaluate factors that limit salmonid production in the Shasta River (Jeffres et al., 2008). The thermal regime of the Shasta River is one aspect of instream habitat and should be considered in conjunction with hydrology, other water quality factors, geomorphology, fish life histories, and human alterations.

The California Nature Conservancy purchased 1,700 acre Nelson Ranch in 2005 with a goal of preserving habitat for anadromous salmon while simultaneously preserving the traditional ranching lifestyle of the Shasta Valley (TNC, 2007). The Nature Conservancy has granted permission for the studies described herein. Approximately five miles of the Shasta River (RM 27.36 - 32.10) can be accessed from Nelson Ranch, which is entirely on the east side of the river (Figure 20). The upstream property boundary of Nelson Ranch is 1.61 river miles downstream of Big Springs, and the GID/Huseman Ditch diversion is located at river mile 30.59 (the confluence of the Shasta River with the Klamath River is river mile 0, and Dwinnell Dam is river mile 40.62). In the Shasta Valley, the irrigation season spans from approximately April 1 to October 1, although exact dates of water rights vary by permit owner. Within Nelson Ranch, the upstream portion of the river is predominantly riffle/run, while the downstream portion is mainly meandering reaches. Since access to the Shasta River from other properties was difficult to secure, the five river-miles of the Shasta River accessible from Nelson Ranch are assumed to be representative of general upstream Shasta River thermal conditions unless otherwise noted.

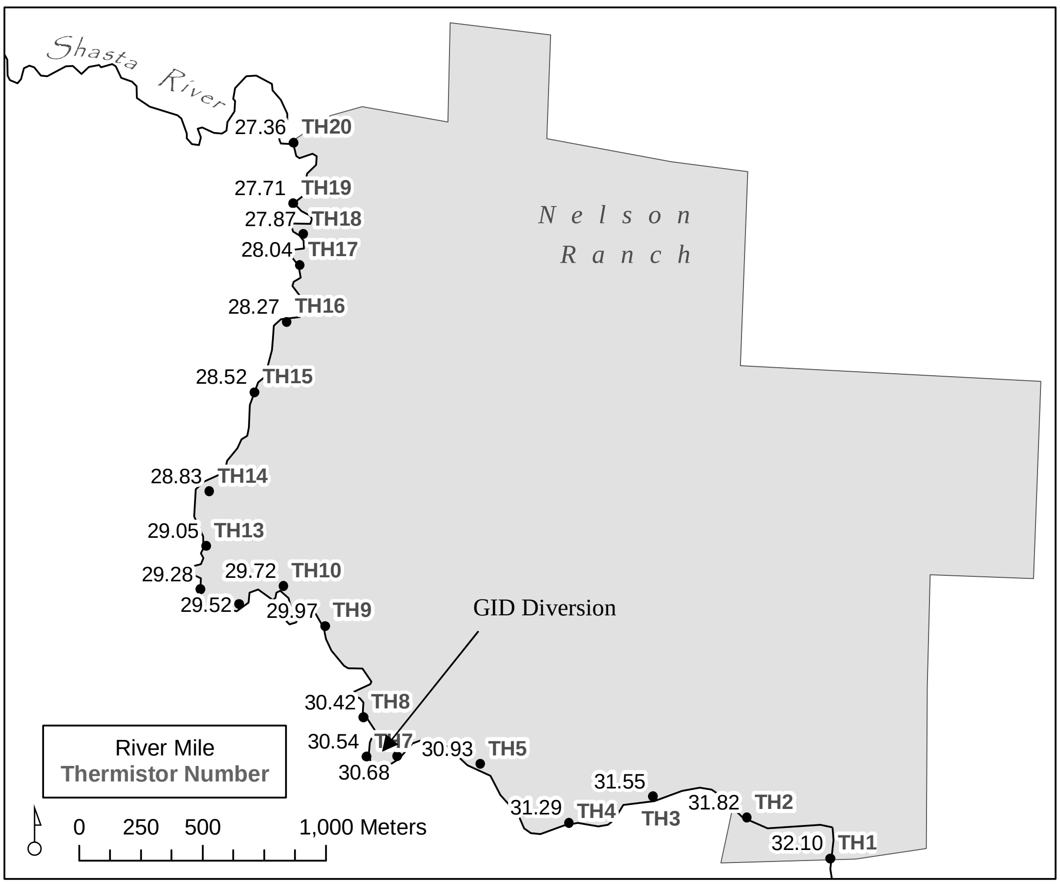


Figure 20. Remote logging thermistor layout and locations by river mile

Longitudinal Temperature Variability

Hourly longitudinal water temperature data was analyzed from 5/20/06 to 5/1/07, the period of record available as of summer 2007. 2006 was a wet year, and 2007 was a dry year. This section begins by analyzing water temperature differences at the upstream and downstream property boundaries using daily mean, maximum, and minimum temperature data, and by comparing total and monthly means using Student's t-tests. Box and whisker plots show hourly and monthly trends at the property boundaries, with emphasis given to differences in timing of daily maximum and minimum temperature peaks. Finally, true longitudinal analysis compares all temperature loggers in the Shasta River along Nelson Ranch during for a representative day in all months. This section concludes with major findings and recommendations for future studies.

Onset StowAway Tidbit temperature loggers were deployed, maintained, and downloaded by the U.C. Davis Center for Watershed Sciences to monitor longitudinal water temperature. Twenty loggers were deployed over the length of the Shasta River on Nelson Ranch (Figure 20). These devices are accurate to +/- 0.2°C within the range of temperatures typically experienced in the Shasta River, with slightly lower accuracy at the low and high ends of their range. Temperature loggers were placed on the bed toward the center of the river where flow provides a well-mixed, representative main channel water temperature. Loggers are numbered from upstream to downstream, so that loggers 1 and 20 are at the upstream and downstream property boundaries, respectively. Due to numerous gaps in recorded data, a temperature pattern for the upstream boundary of Nelson Ranch was created by combining the uppermost three loggers to make a nearly continuous upstream boundary temperature series, and by combining the four lowermost loggers for a downstream boundary temperature series (see Appendix A for a table of logger completeness).

Overall, the Shasta River at the upstream portion of Nelson Ranch had greater thermal variability than the downstream portion, particularly during summer (Figure 21 - Figure 24) (see Appendix A for monthly figures of hourly water temperature with air temperature and instream flow). On average, water temperature was 0.2°C cooler near the upstream boundary than the downstream boundary (Figure 22), indicating atmospheric heating through the Nelson Ranch reach.

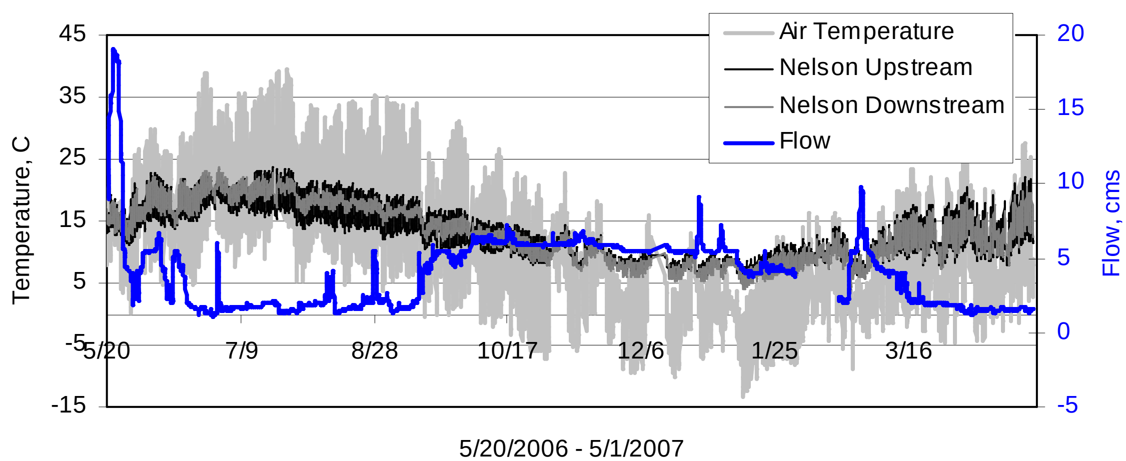


Figure 21. Hourly water temperature, air temperature, and instream flow at Nelson Ranch property boundaries

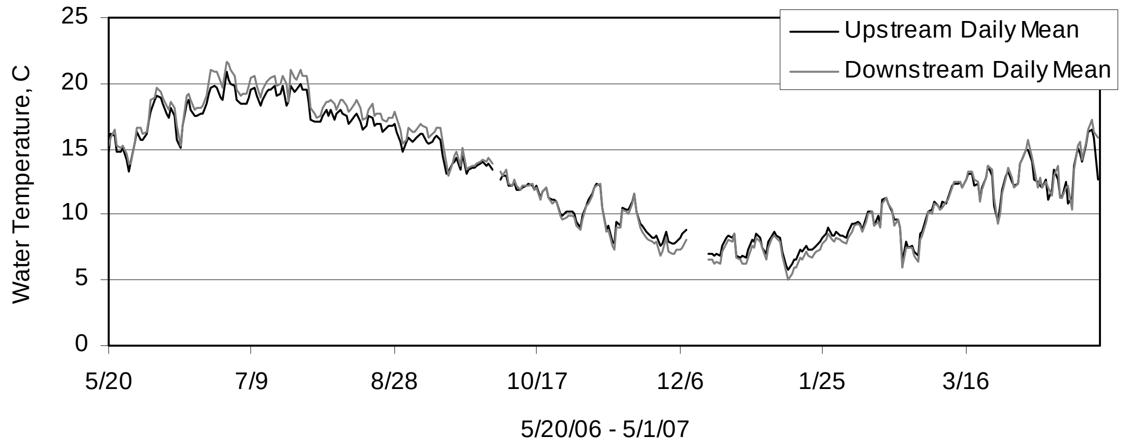


Figure 22. Daily mean water temperature at Nelson Ranch property boundaries

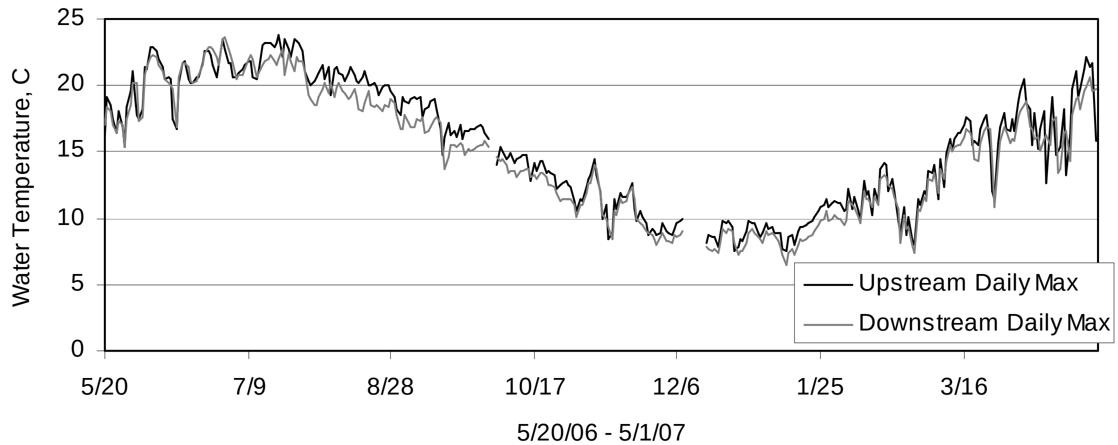


Figure 23. Daily maximum water temperature at Nelson Ranch property boundaries

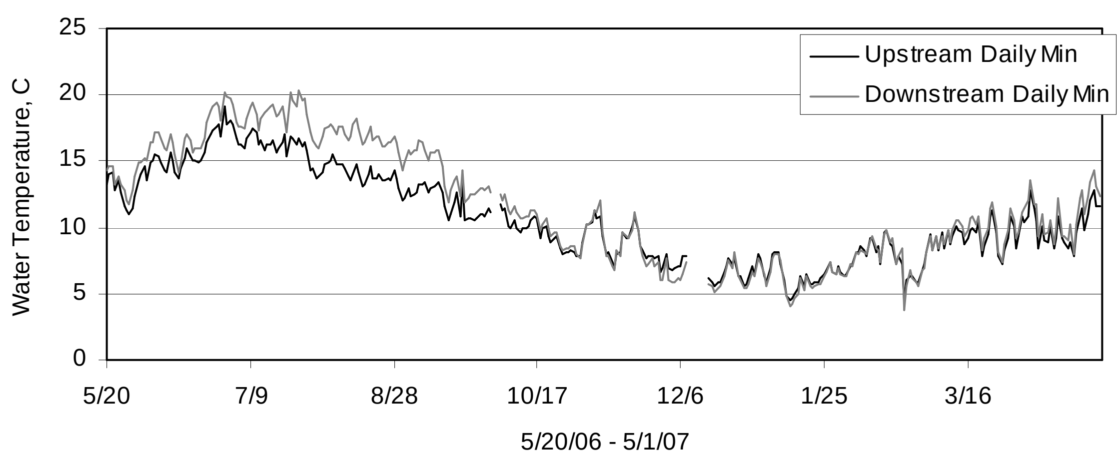


Figure 24. Daily minimum water temperature at Nelson Ranch property boundaries

Two-sample t-tests were used to compare whether the total and monthly means in water temperature between the upstream and downstream boundaries of Nelson Ranch were statistically significant. Due to the travel time necessary to carry pulses of water down the Shasta River, water temperature data from the upstream and downstream property boundaries of Nelson Ranch were assumed to be independent, rather than paired at given times. Travel time changes with flow volume, thus lagged pairs were also not used. Two-tailed T-tests were computed using SYSTAT for Windows v.11.

Using a 95% confidence level, mean water temperature between the upstream and downstream boundaries of Nelson Ranch was statistically different using the entire data set, and for monthly subsets excluding October, March, and April (Table 7). Where t-test results imply that the upstream and downstream water temperature means differed (p-value < 0.05), there is a 5% chance that the true difference between the temperature means falls outside of the confidence intervals listed in Table 7. Probability mass functions illustrate the differences between the mean and variance of monthly water temperature at the upstream and downstream boundaries of Nelson Ranch (Figure 25 - Figure 27). Overall, annual temperature variability is greater at the upstream boundary than the downstream boundary. During summer, mean water temperature increases by more than 0.5°C between the upstream and downstream property boundaries of Nelson Ranch, although thermal variability decreases at the downstream boundary. Reduced thermal variability near the downstream property boundary may occur from warm upstream pulses of water arriving at the lower property boundary at night when solar heating is absent, and will be discussed further in the following sections.

Table 7. T-test means, mean differences, confidence intervals, degrees of freedom, and p-values

	Upstream		Downstream		Mean	Confidence		
	Mean	SD	Mean	SD	dif.	Interval	df	p-value
All	***12.8***	***4.4***	***13.0***	***4.6***	***-0.2***	***-0.3 to 0.0***	***16304***	***0.029***
May	15.0	2.2	15.4	1.8	-0.4	-0.7 to –0.1	600	0.014
Jun	18.0	2.3	18.7	2.1	-0.6	-0.9 to –0.4	1438	0.000
Jul	19.1	2.2	20.0	1.3	-0.9	-1.1 to –0.7	1486	0.000
Aug	17.1	2.3	17.8	1.0	-0.8	-1.0 to –0.6	1486	0.000
Sep	14.6	2.2	15.1	1.4	-0.5	-0.6 to –0.3	1438	0.000
Oct	11.6	1.8	11.6	1.5	0.0	-0.2 to 0.2	1424	0.787
Nov	9.7	1.6	9.5	1.7	0.3	0.1 to 0.4	1438	0.002
Dec	7.7	1.1	7.2	1.0	0.5	0.4 to 0.6	1220	0.000
Jan	7.7	1.4	7.3	1.3	0.4	0.3 to 0.6	1486	0.000
Feb	9.1	1.8	8.8	1.7	0.2	0.1 to 0.4	1342	0.015
Mar	11.6	2.5	11.6	2.2	0.0	-0.2 to 0.3	1486	0.904
Apr	13.3	3.3	13.5	2.6	-0.2	-0.5 to 0.2	1438	0.319

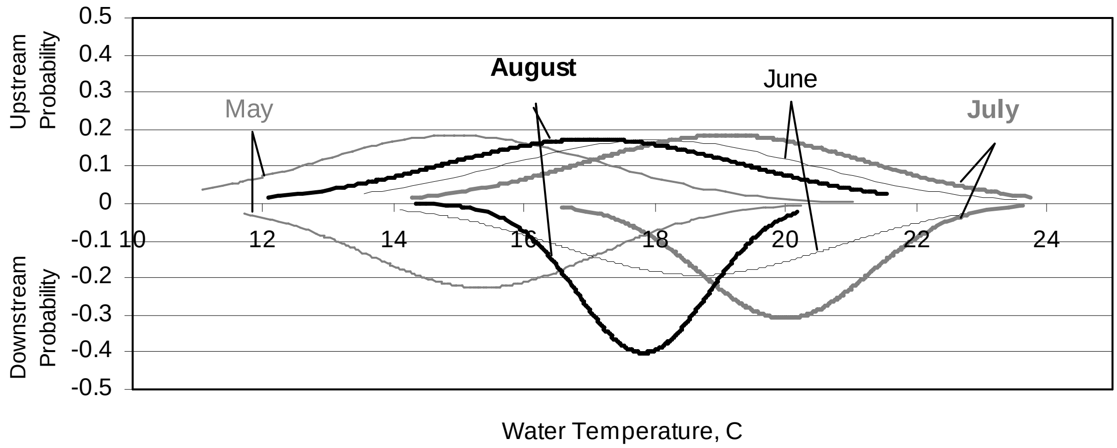


Figure 25. May -August water temperature probability mass functions

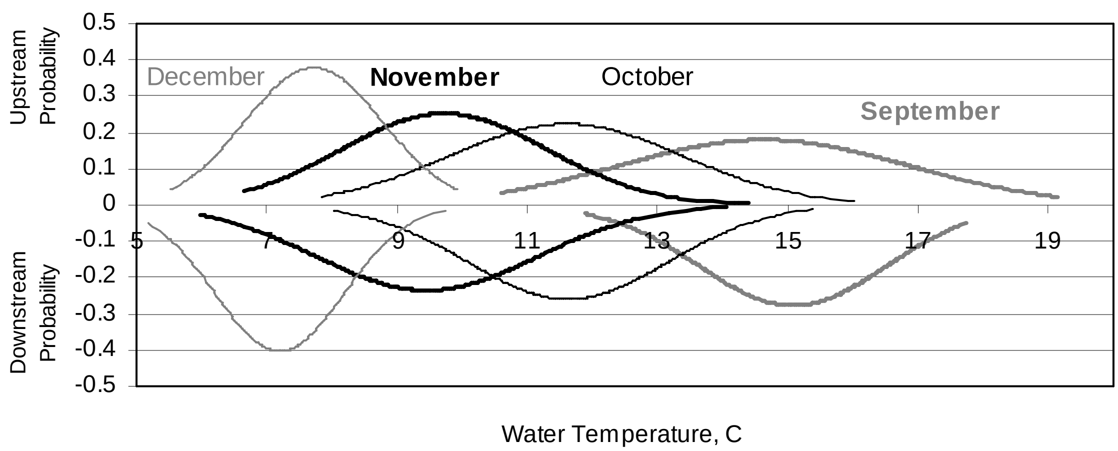


Figure 26. September - December upstream and downstream water temperature probability mass functions

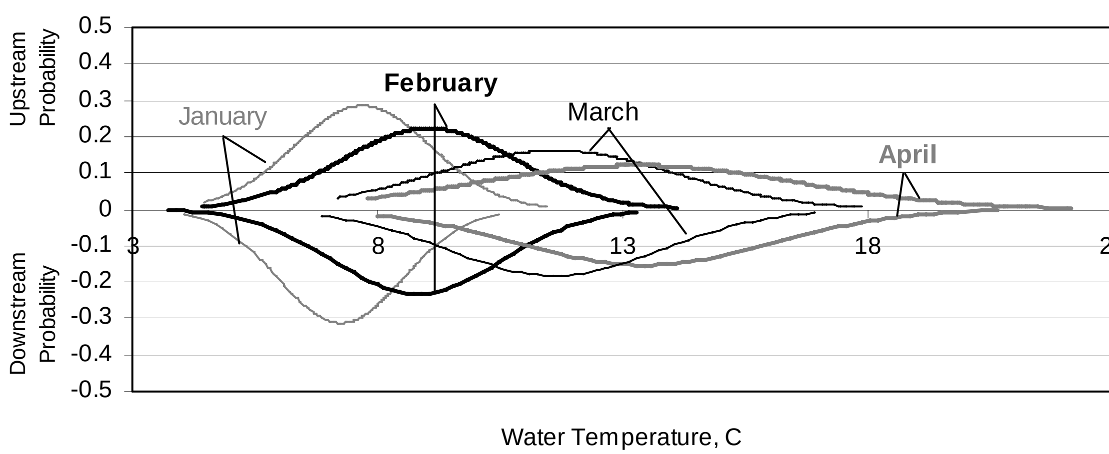


Figure 27. January - April upstream and downstream water temperature probability mass functions

Hourly and monthly analysis of water temperatures at the upstream and downstream boundaries of Nelson Ranch show more substantial temperature differences than the t-tests. Hourly water temperature box and whisker plots from the upstream boundary of Nelson Ranch (Figure 28) are typical of a California river. Maximum water temperature occurs between 5:00 – 6:00 pm, except during the long days of summer when maximum water temperature can occur as late as 8:00 pm. Minimum water temperature occurs in early morning, between 7:00 – 9:00 am.

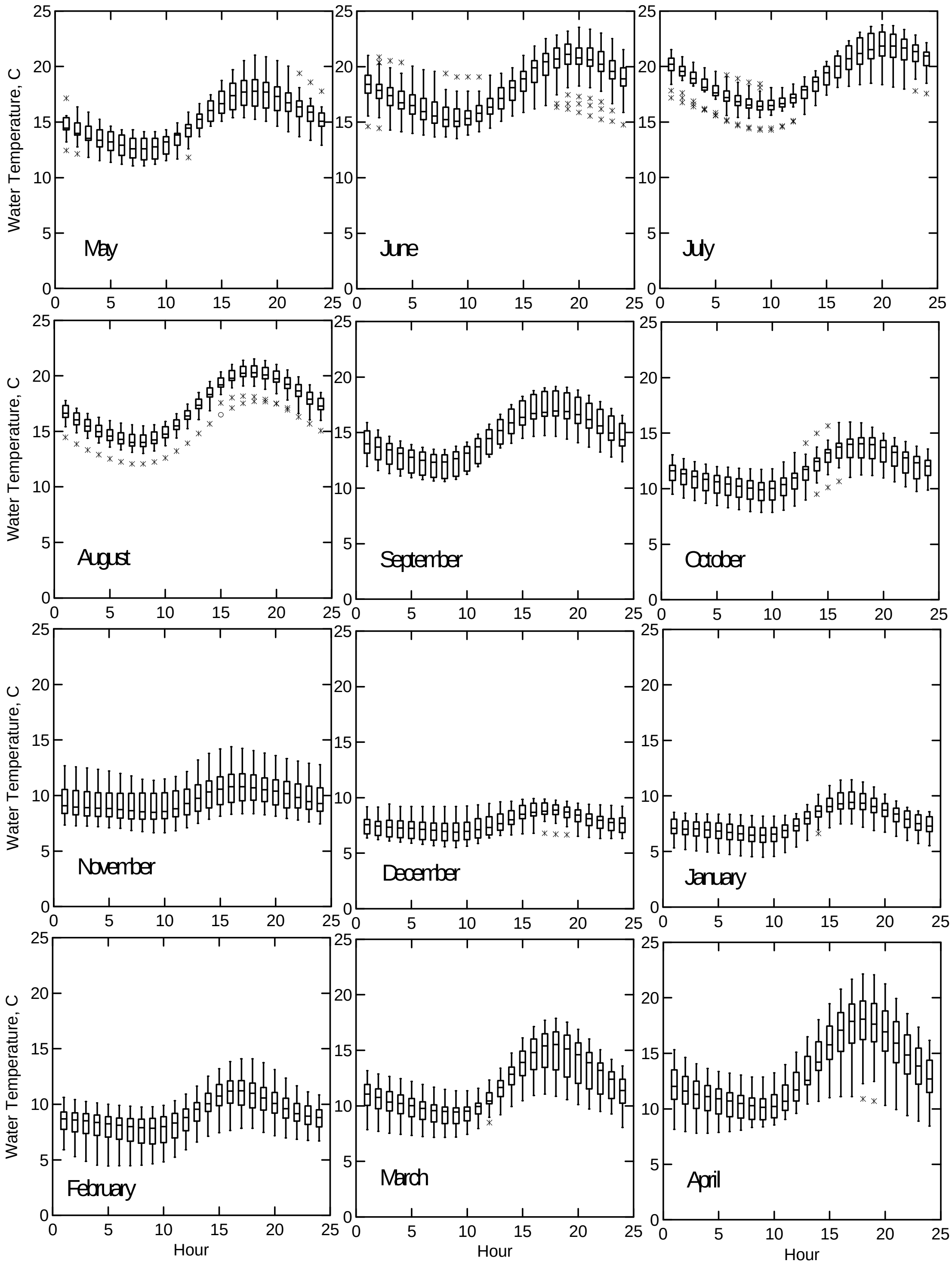


Figure 28. Hourly water temperature variability by month at Nelson Ranch upstream boundary

One of the more interesting findings of this dataset was the discovery of nightly water temperature peaks near the downstream property boundary of Nelson Ranch (Figure 29). At that location, maximum water temperature peaks between 9:00 pm and 4:00 am in all months. Minimum hourly water temperature occurs a few hours later than at the upstream boundary, between 8:00 – 11:00 am. Although October, March, and April had little difference in monthly means, all months show clear thermal differences on an hourly scale (Figure 28 and Figure 29). The dominant thermal influence for the Shasta River is daytime solar radiation and advection of thermal energy from upstream sources. This implies that a volume of warm water, originating upstream of Nelson Ranch, is being transported downstream and is reaching Nelson Ranch's downstream boundary at night. A unique thermal signature with two daily peaks was also observed toward the downstream property boundary of Nelson Ranch, most likely from a combination of warm water originating above Nelson Ranch arriving at the downstream boundary at night combined with daily solar radiation (Figure 30).

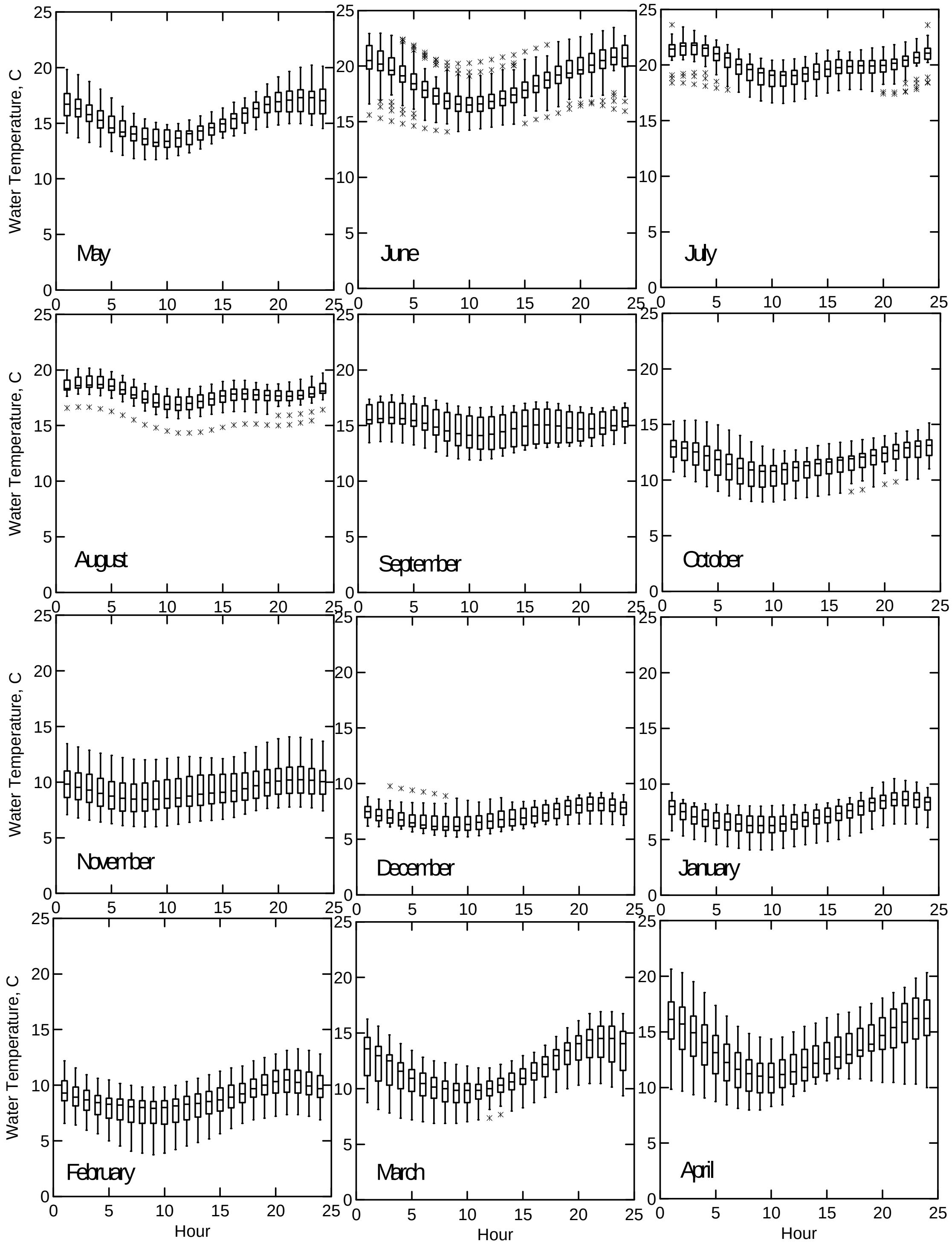


Figure 29. Hourly water temperature variability by month at Nelson Ranch downstream boundary

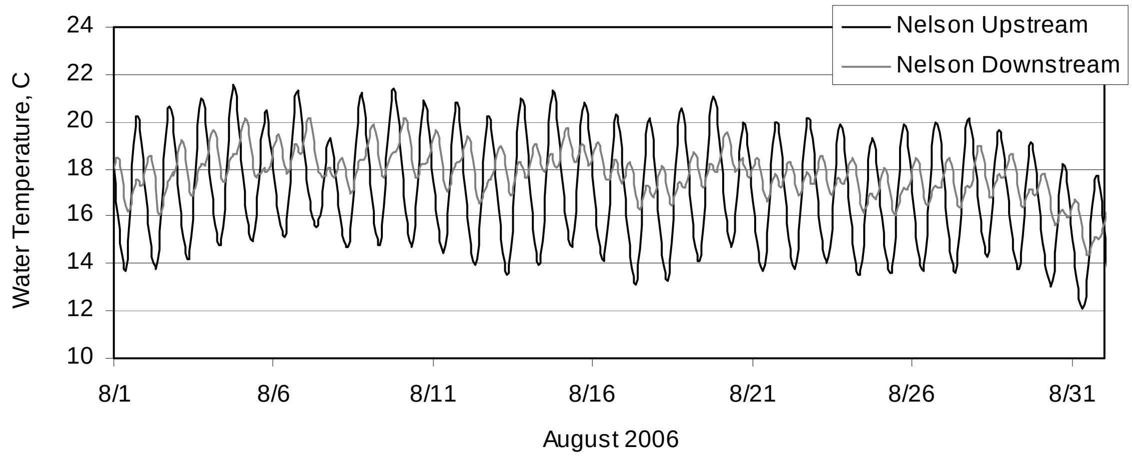


Figure 30. August water temperature at Nelson Ranch property boundaries

The time of maximum daily temperature at the upstream property boundary occurs approximately four hours earlier than at the downstream boundary, except July through mid-October when the time between daily upstream and downstream maximum temperature widens to approximately eight hours. (Table 8, Figure 31). Data from the downstream boundary show a marked difference in the timing of temperature peaks and valleys during mid-July to mid-October, and the rest of the year. This change in timing has to do with a combination of low flow conditions increasing travel time in the river, warm water inflows from tailwater return, or other undescribed inflows, such as from the Big Springs complex upstream of Nelson Ranch. There is considerable variability in the times of the daily maximum water temperature during all months. The time of minimum daily water temperature is similar between the up and downstream boundaries of Nelson Ranch, except mid-July to mid-October when the upstream minimum water temperature occurs approximately three hours earlier than at the downstream boundary (Figure 32). Assuming a rectangular channel with a width of 40 ft (12.2 m), depth of 2 ft (0.6 m), distance of 5 mi (26,400 ft) (8,047 m), and average velocity of 111 cfs in mid-July to mid-October, and 145 cfs for the rest of the year, travel time increases from 4 hours for the majority of the year to 5 hours and 20 minutes from July to October, showing travel time through Nelson Ranch increases during summer.

Table 8. Monthly mean, maximum, and minimum water temperature (C) at upstream and downstream property boundaries, with average times of maximum and minimum temperatures.

	Upstream					Downstream				
	Mean	Max	Min	Mean max temperature hour	Mean min temperature hour	Mean	Max	Min	Mean max temperature hour	Mean min temperature hour
May	15.06	21.02	11.06	18:00	8:00	15.33	20.22	11.71	22:00	9:00
June	18.02	23.53	13.53	19:00	9:00	18.66	23.47	14.10	23:00	10:00
July	19.10	23.76	14.29	20:00	9:00	19.99	23.62	16.56	3:00	11:00
August	17.05	21.53	12.07	18:00	7:00	17.81	20.17	14.34	4:00	11:00
September	14.63	19.15	10.59	18:00	7:00	15.08	17.75	11.88	3:00	11:00
October	11.60	16.01	7.85	18:00	9:00	11.62	15.39	8.05	24:00	9:00
November	9.72	14.39	6.66	17:00	9:00	9.46	14.07	5.98	21:00	8:00
December	7.72	9.93	5.51	18:00	9:00	7.20	9.76	5.18	22:00	8:00
January	7.69	11.44	4.48	17:00	9:00	7.25	10.47	4.06	22:00	8:00
February	9.04	14.10	4.43	17:00	9:00	8.81	13.27	3.74	21:00	9:00
March	11.62	17.89	7.17	18:00	9:00	11.61	16.91	6.88	22:00	10:00
April	13.30	22.15	7.80	18:00	9:00	13.46	20.64	7.98	23:00	10:00

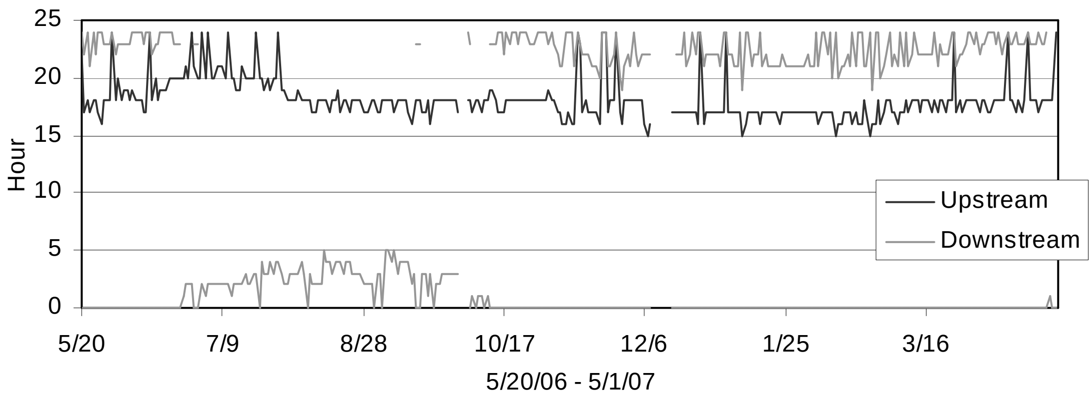


Figure 31. Hour of daily maximum water temperature

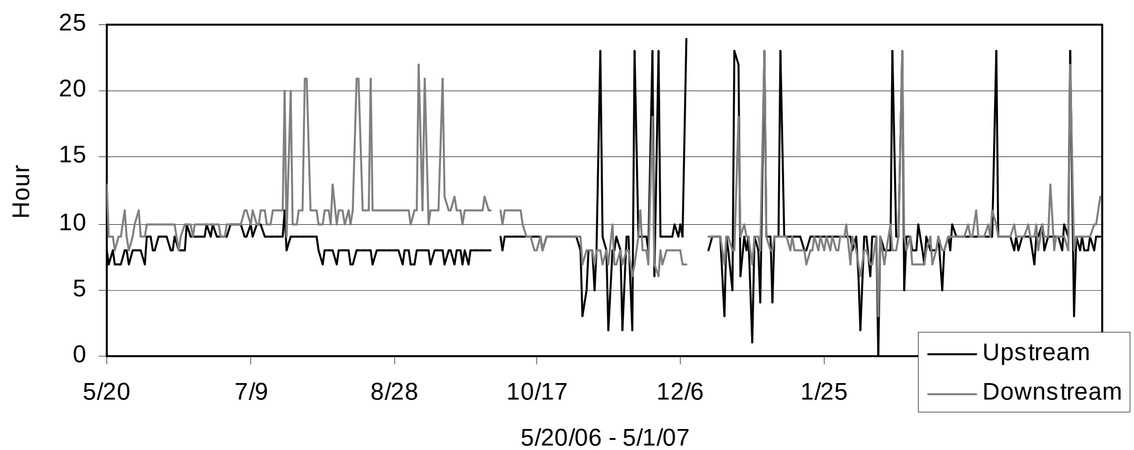


Figure 32. Hour of daily minimum water temperature

To understand how thermal patterns change along the Shasta River, box and whisker plots display longitudinal water temperature through Nelson Ranch (Figure 33) during a representative day in the middle of each month to highlight spatial and temporal changes in the Shasta River. The Grenada Irrigation District diverts water to the Huseman Ditch at river mile 30.59, between loggers 6 and 7 (2628 m and 3066 m from the upstream boundary of Nelson Ranch, respectively). Heating in the GID diversion pond could be expected; however, there are no clear temperature trends from the GID diversion during irrigation season. Likewise tailwater from Nelson Ranch returns to the Shasta River between loggers 1 and 2 (438 m and 836 m from the upstream boundary, respectively). During irrigation season, data from both these loggers only exists during the August and September plots, and no clear trend is present.

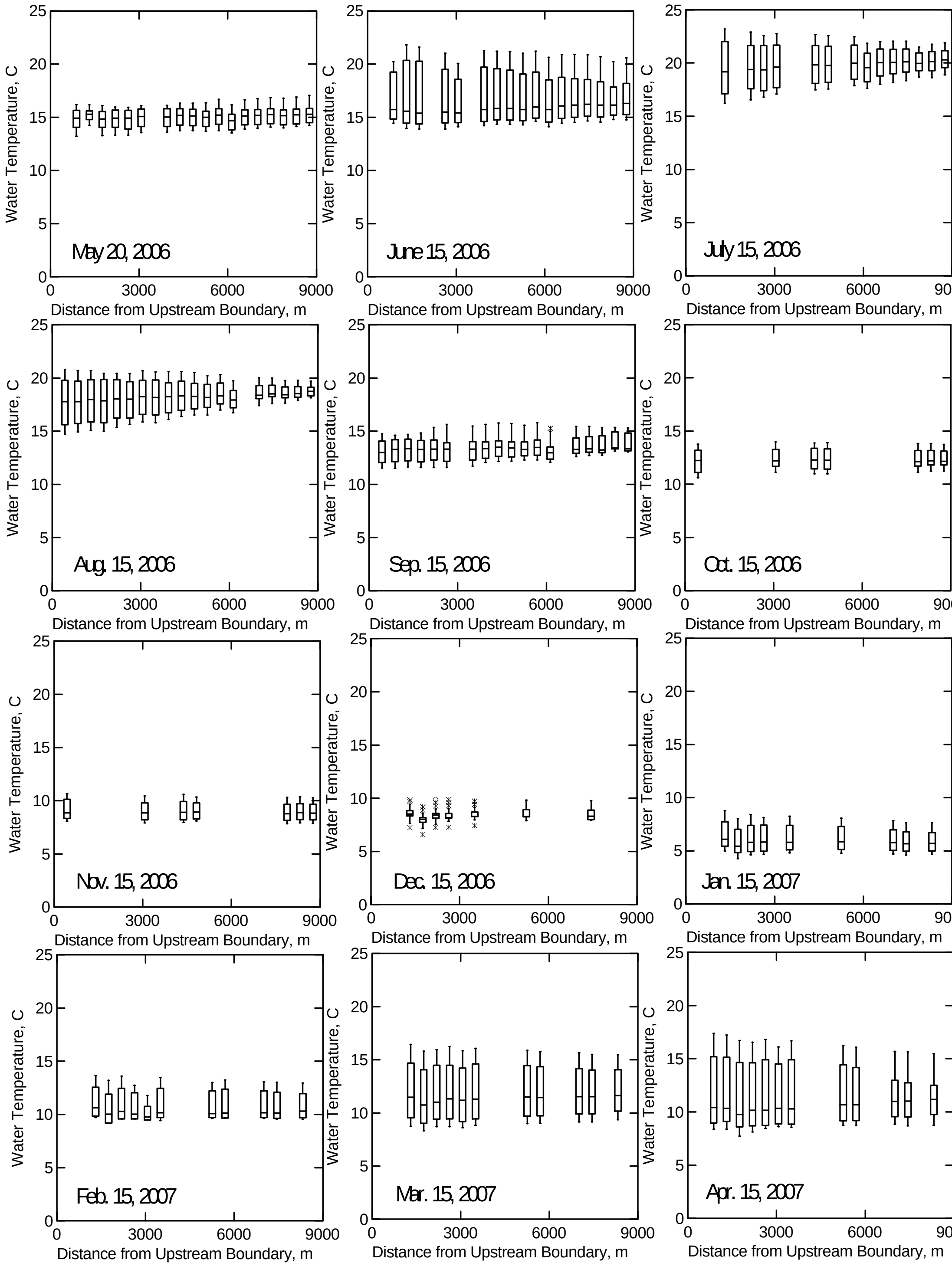


Figure 33. Mid-monthly longitudinal box and whisker plots along Nelson Ranch

Longitudinal Analysis Conclusions

A year of continuous longitudinal temperature data enables thorough analysis of temporal and spatial thermal conditions in the Nelson Ranch reach of the Shasta River. Previous monitoring efforts have sampled water temperature at widely spaced locations. Therefore, this research helps illuminate small-scale thermal conditions and variability not evident in previous studies. Several interesting trends were observed. From June to August, mean water temperature rises with distance from the upstream property boundary; although during these months, and also in March and April, there is less thermal variability toward the downstream boundary.

Discovery and documentation of nightly temperature peaks at the downstream property boundary of Nelson Ranch is another interesting observation. The dominant thermal influence for the Shasta River is solar radiation, which is absent after sunset. This implies that a volume of warm water is being transported downstream and is reaching Nelson Ranch's downstream boundary at night. The thermal condition at the upstream boundary of Nelson Ranch is inherited from upstream factors including springflow contribution from Big Springs (and other sites), the upper Shasta River, Parks Creek, and human factors, such as diversions (including diversion ponds) and agricultural return flows. The data from the longitudinal temperature loggers deployed at Nelson Ranch are insufficient to determine the cause of the incoming warm water. Further study is recommended to determine the source and volume of possible warm inflows upstream of Nelson Ranch.

Exploring Local Thermal Diversity

Exploratory temperature probing and lateral river transects were conducted to improve understanding of small-scale thermal diversity in the Shasta River. This section begins with a description of exploratory temperature probing conducted during summer 2006 to identify possible cool-water habitat from small springs, subsurface flows, or seeps to the Shasta River (site illustrations and detailed data are in Appendix A). Also during summer 2006, water temperature was recorded in three lateral transects to assess the extent of heating near the shallow margins of riverbanks. This section ends with analysis of temperature transects deployed during winter and spring to assess thermal diversity during these seasons. These measurements are important to understand local differences in habitat that are too small to be apparent in modeling studies.

Seven sites were sampled for thermal diversity on Nelson Ranch property from 8/22/06 - 8/23/06. In addition, cross sectional water temperature was measured along three transects in the Shasta River near the Nelson Ranch return flow ditch on 8/22/06 (Figure 34). Water depth was measured with a Global Water pressure transducer (model WL 16) accurate to +/-0.2% in the 0-21°C range. A Tech Instrumentation model TM99A temperature unit with a model 2007 probe was used for most handheld temperature sampling. The TM99A temperature unit is accurate to +/- 0.1°C in the 0-40°C range. An Oakton Acorn Series Temp 5 handheld temperature unit with a 15 cm steel probe was used to assess bed temperature by inserting the probe into the bed matrix to assess pore water temperatures (and river temperature when noted). The Oakton Acorn temperature probe is accurate to +/- 0.2°C. Mechanisms for exchange of pore water with surface water were not analyzed.

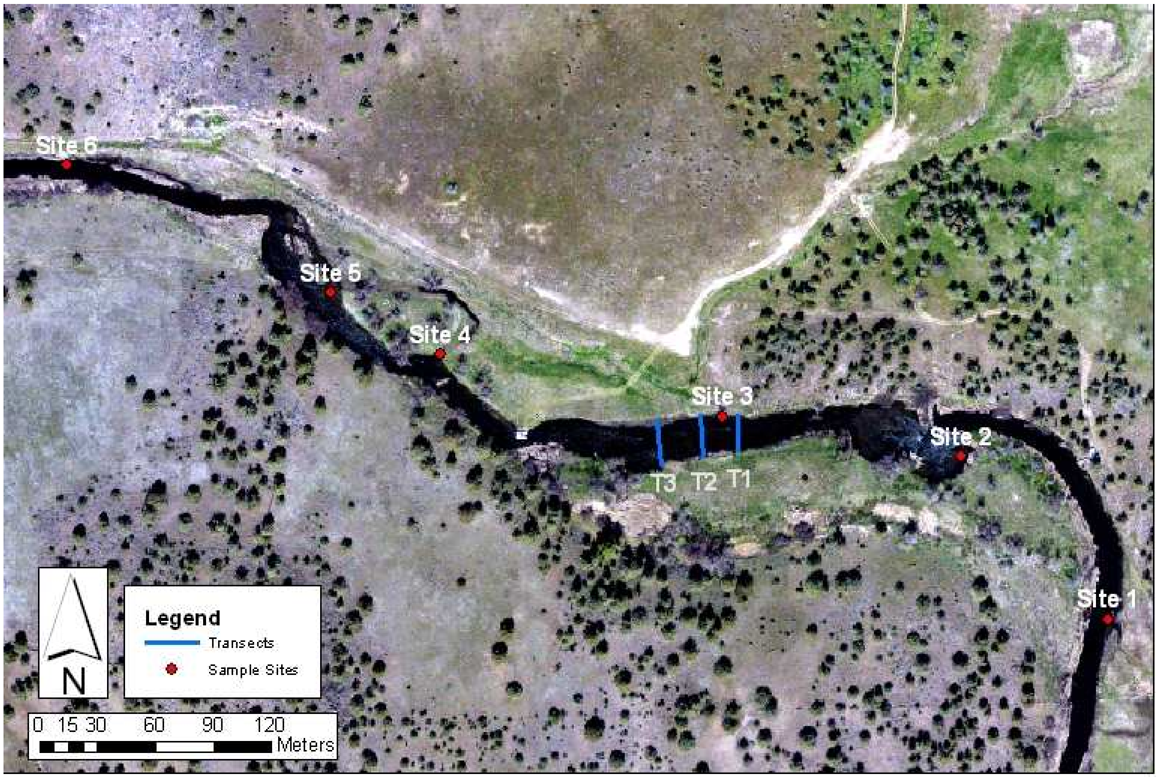


Figure 34. Temperature probing locations, August 2006

The pressure transducer and TM99A temperature unit were mounted to Plexiglas on a 1.8m rod. Probe tips were attached to the end of the rod, so temperature and depth measurements could then be taken simultaneously in water up to 1.5m. The handheld device allowed quick assessment of vertical distribution of water and bed temperature, with the ability to explore under overhanging vegetation or cutbanks.

Temperature observations throughout the year identified several key insights into smaller scale thermal conditions on the Nelson Ranch. During summer, small, localized cool water refugia associated with subsurface flow, seeps, and/or springs were identified on the Nelson Ranch with temperatures up to 1-2°C cooler than mainstem river conditions. Irrigation return flows also occur on the Nelson Ranch. However, both cool refugia and return flows were generally small in size and/or magnitude, and did not appear to have an appreciable influence on overall mainstem temperatures.

Although local water temperature differences are small (<1-2°C), they may improve instream conditions for fish and other wildlife, particularly when water temperature nears critical limits for cold-water fish species. Additional field observations are needed to adequately quantify small-scale water temperature changes at the sites discussed above, and to identify whether cool water originates from springs, subsurface flow, shading, or local channel characteristics (particularly shallow channels that heat and cool faster than the Shasta River). Alternative management strategies to reduce heating of cool water sources warrant additional field sampling and data analysis.

Shasta River Lateral Variability

The Shasta River was surveyed in three cross-sectional transects on 8/22/06 to provide water temperature measurements testing the occurrence and extent of heating at river margins. Installation of six lateral transects in the mainstem Shasta River and two longitudinal profiles in side channels then recorded hourly water temperature from late January to late June, 2007. The January to June transects tested whether channel margins and side channels have greater thermal variability than the mainstem, and whether floodplain bench habitat increases the variety of thermal conditions available for habitat. This section outlines methods and findings of the instantaneous cross-sections completed during summer 2006, followed by discussion of major findings of the winter and spring transects (See Appendix A for a description of winter and spring lateral transect sites with noting location and habitat characteristics).

Summer 2006 – Instantaneous temperature cross-sections

Water temperature was sampled across the Shasta River in transects to explore whether heating occurs along riverbank margins, and the extent of heating. Sampling was completed on 8/22/06 using a pressure transducer and temperature probe (described above). The uppermost transect was located upstream of the Nelson Ranch return flow channel, and the middle and lower transects were downstream of the channel. All three transects were above the CDFG screw trap (Figure 34, Figure 35). Locations of transects were chosen as likely to capture margin warming and where the river was shallow enough to wade. Low herbaceous vegetation was present on both banks, representative of the reach.

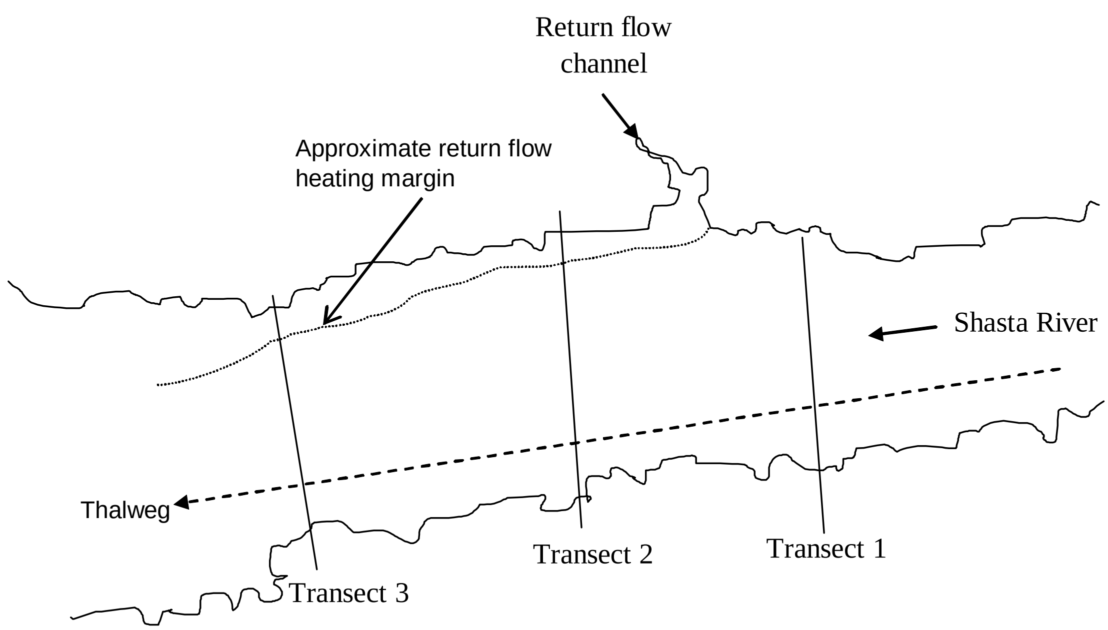


Figure 35. Summer transect overview sketch

There was slight margin warming in transect 1 above the Nelson Ranch return flow ditch (Figure 36 & Figure 37). Transect 2 shows considerable warming on river right from the return flow ditch, and a slight temperature decrease on river left, possibly from vegetative shading (Figure 38 & Figure 39). Transect 3 was approximately 20-25m below the return flow ditch, and shows more dispersed warming on river right from tailwater return (Figure 40 & Figure 41). Water temperature mid-channel of each transect was vertically and laterally constant. Water temperature was 15.7°C mid-channel of the first transect, 16.3°C in the second, and 17.0°C in the third. This longitudinal temperature increase is an artifact of the time lapse between transects, which were conducted at 12:45, 1:20, and 2:10 respectively.

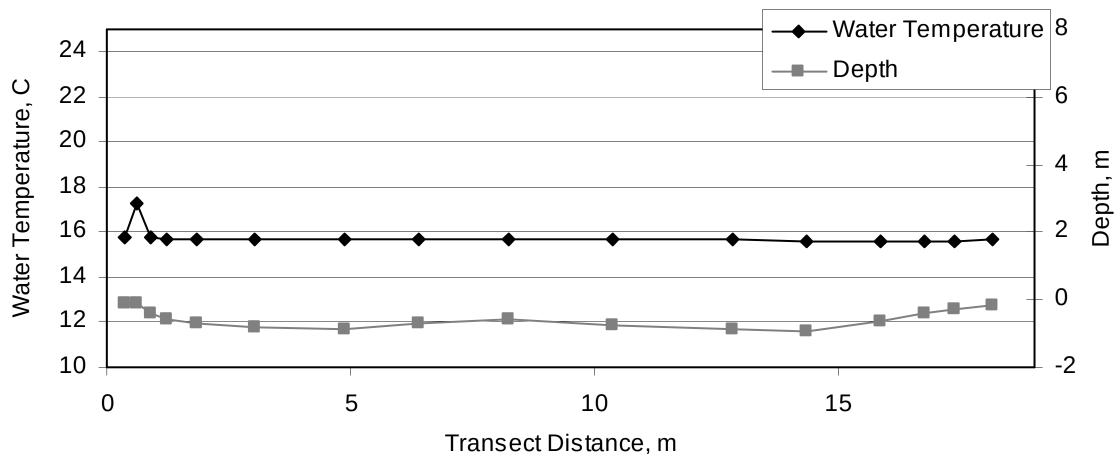


Figure 36. Transect 1 water temperature and depth

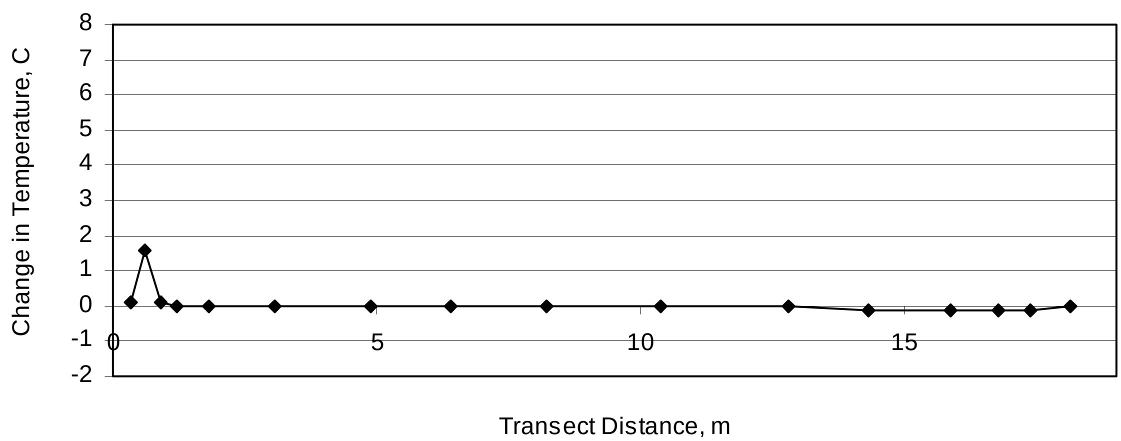


Figure 37. Transect 1: change in temperature from center of river

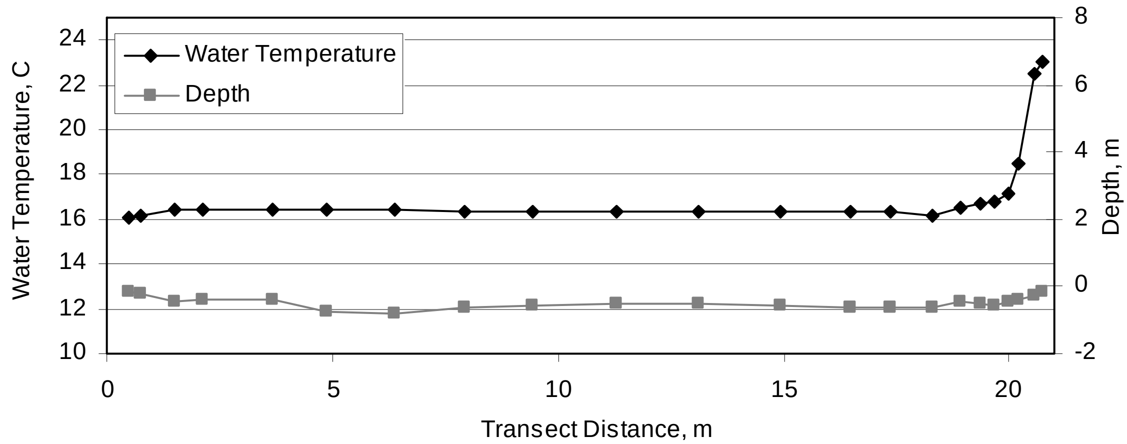


Figure 38. Transect 2 water temperature and depth

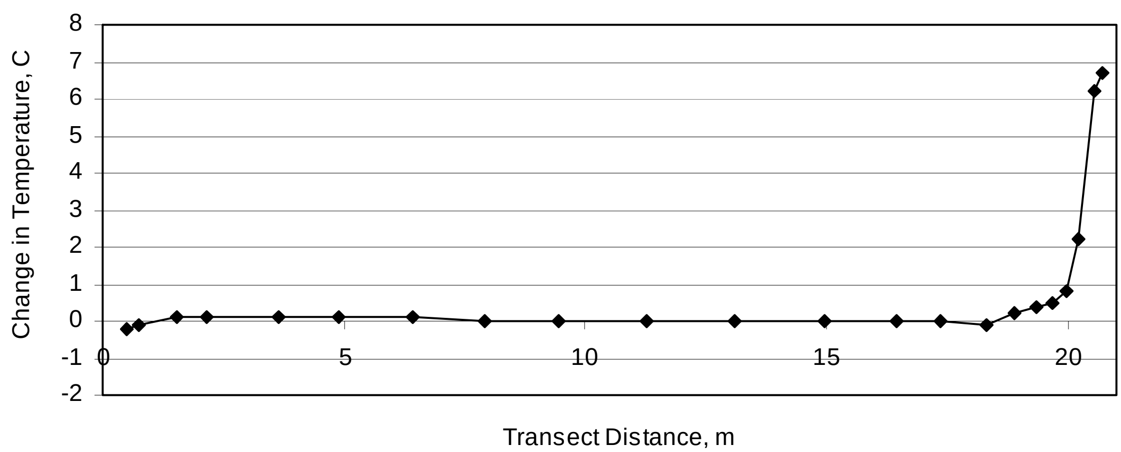


Figure 39. Transect 2: change in temperature from center of river

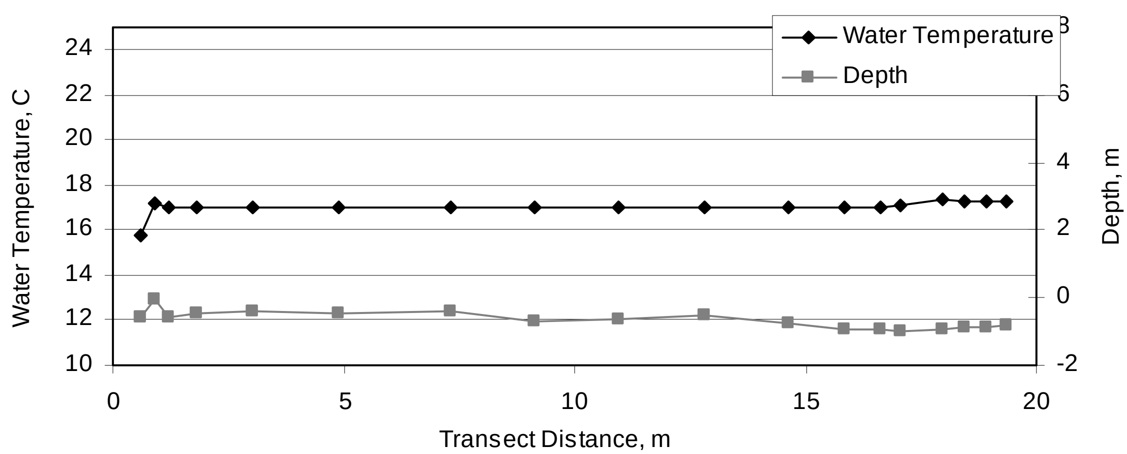


Figure 40. Transect 3 water temperature and depth

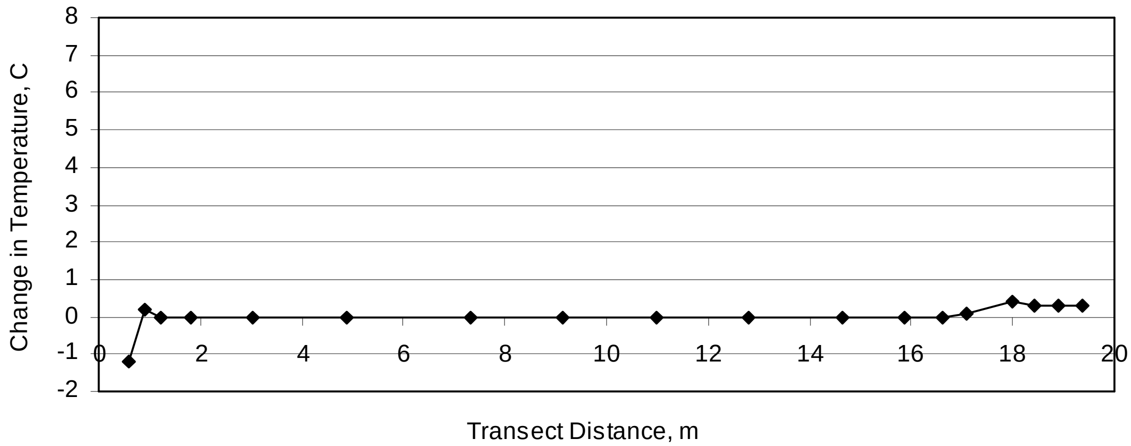


Figure 41. Transect 3: change in temperature from center of river

Winter and Spring Lateral Transects

To further understanding of lateral temperature variability and to characterize baseline winter and spring conditions of the Shasta River along TNC's Nelson Ranch, six lateral transects were installed on 1/28/07. Four were removed on 6/22/07 (Figure 42); although the upper and lowermost transects remained in the Shasta River measuring water temperature through summer 2007. Those transects were not removed because all loggers remained submerged despite low river stage, they did not have other problems (such as sedimentation), and were easy to access. Transects were used to monitor water temperature at floodplain bench, deep channel, and shallow channel habitat types, with two transects at each of habitat type.

Time series of spring and winter transect temperature data are analyzed by comparing maximum daily water temperature at all transect positions. Water temperature differences are tested for statistical significance using two-way ANOVA. In general, logger position in temperature transects is a poor predictor of water temperature, indicating margin heating does not lead to a statistically significant difference in water temperature. Regardless, analysis of daily maximum water temperatures reveal slight differences in thermal variability within transects. Transects in deep water habitat show little thermal variability, transects in shallow habitat have more variability especially during summer months, and transects in floodplain bench habitat were difficult to analyze since loggers became exposed to air by March or April. Temperature loggers were also installed longitudinally in two side channels; however, side channels were frozen until March and were dry by April. Therefore, side channel monitoring provided little, except to conclude that during dry years, side channels on Nelson Ranch may not provide useable habitat for fish.

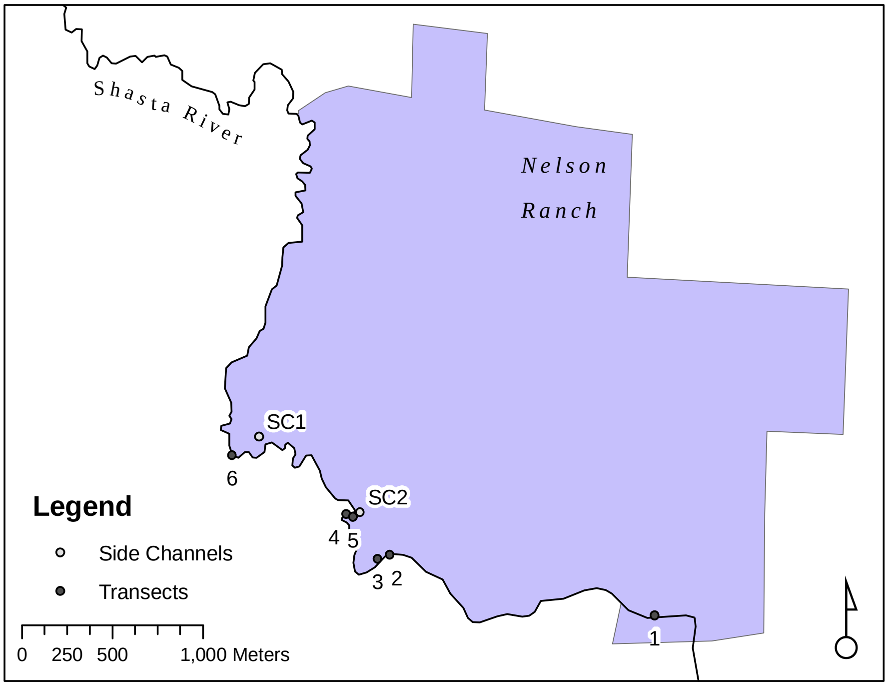


Figure 42. Winter temperature transects and side channel monitoring locations

All transects were located on river right, and extend laterally to where the river was well mixed. Hobo Water Temp Pro and Hobo Water Temp Pro V2 thermistors, manufactured by Onset Computer Corporation, were deployed on 1/28/07. Both thermistor models are accurate to +/- 0.2°C. Mid-channel or deep water thermistors were placed on the bed connected to the bank by a cable leash, and were protected by a neoprene boot. During deployment of temperature loggers, secondary water temperature was measured with an Oakton Acorn Series Temp 5 handheld temperature unit, accurate to +/- 0.2°C. Water depth was measured with a measuring tape. Transect 1 is included here as an example, illustrations and temperature data for the remaining five transects are in Appendix A.

Transect 1 was located above the CDFG screwtrap site and below Nelson Ranch's agricultural return flow channel. For these purposes, it was considered deep habitat because the right bank is steep (approximately vertical), with little difference in depth to promote margin warming (Figure 43). Logger 1-1 was connected to a 3m cable and placed in the well-mixed portion of the channel. Loggers 1-2 and 1-3 were deployed on the bed, secured by 0.5m cables to the river bottom. Logger 1-4 was deployed under a submerged cutbank in the river bank.

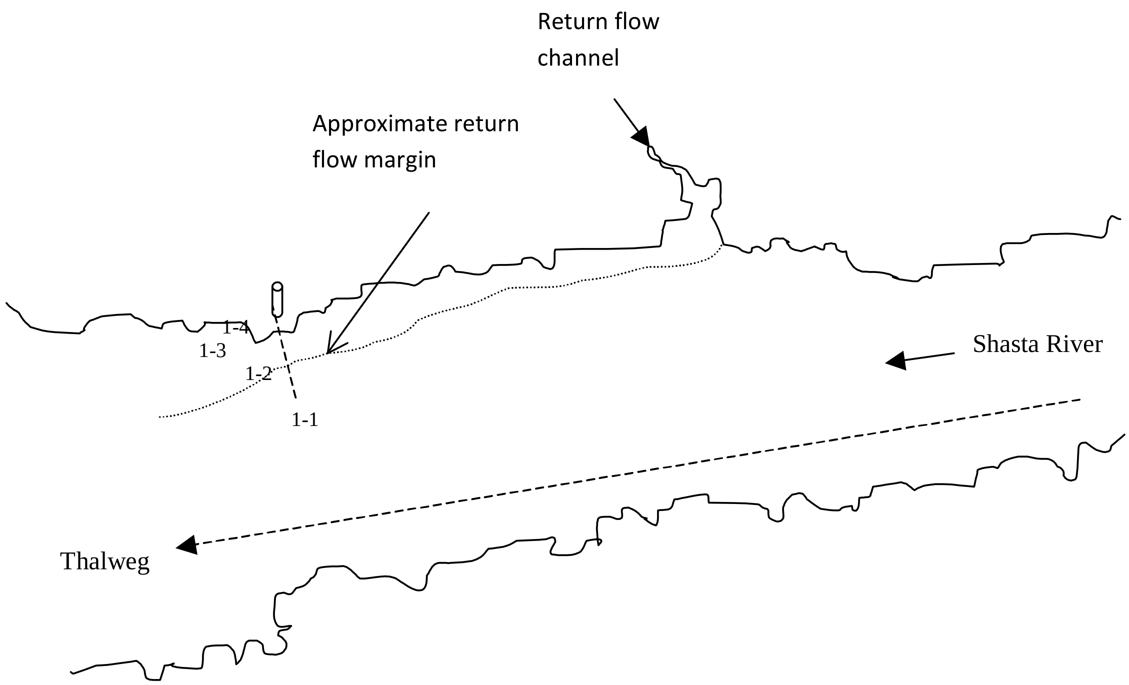


Figure 43. Transect 1 – Schematic of shallow channel habitat above screwtrap site

The warmest and coolest daily maximum water temperature within transects were compared to illustrate small thermal differences. At transect 1, the logger placed mid-channel showed the least thermal variability, and temperature signals from the other three loggers were nearly identical. Recorded water temperature was sensitive to download periods. Possibly aquatic and/or riparian vegetation was disturbed during downloads leading to gradually widening maximum temperature differences (Figure 44).

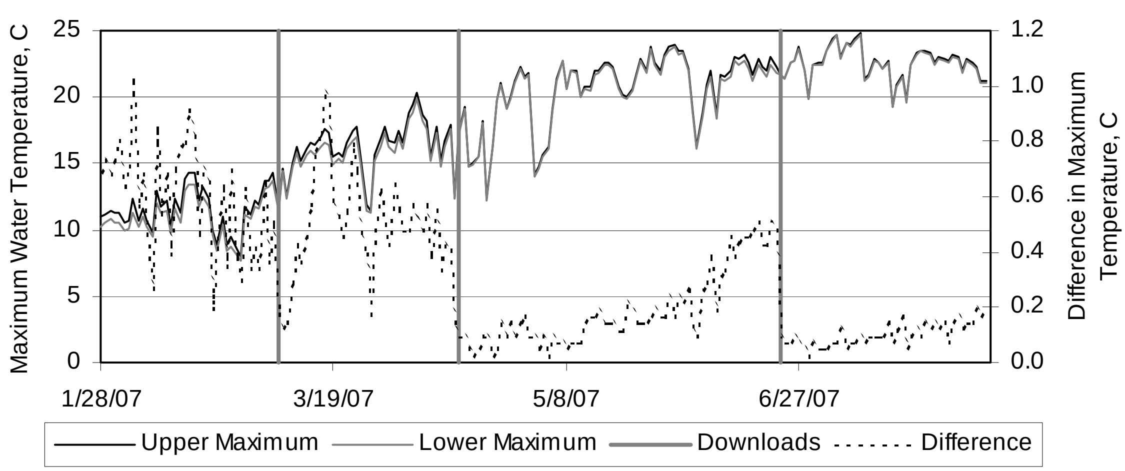


Figure 44. Transect 1 upper, lower, and difference in maximum daily water temperature

Analysis of Winter and Spring Temperature Transect Data

Two-way analysis of variance (ANOVA) was used to test whether mean water temperature at different positions within all transects were statistically significant, using SYSTAT v.11 for Windows. ANOVA is a general technique to test the hypothesis that means among two or more groups are equal, and assumes sample populations are normally distributed and have equal variance. Two-way ANOVA uses two factors, or categorical predictor variables. Here, water temperature was the dependent variable, and the two independent factors were logger position along transects and month. This experimental design does not directly analyze temperature differences between different transects, which were deployed in different habitat types. Loggers in some transects were exposed to air, making different sample sizes at different transects. Thus a method to simultaneously analyze water temperature between transects, as well as within transects (such as repeated measures ANOVA) was not used. Transects 1 and 3 were deployed in deep-water habitat, 2 and 5 were in floodplain bench habitat, and 4 and 6 were in shallow water habitat.

Month was a statistically significant predictor of water temperature at all transects. Water temperature varies greatly throughout the year based on season and weather conditions (Table 9). (Note: Transects 2 and 5, both deployed in floodplain bench habitat, were exposed to air. Results for these transects are broken into two periods: when there are five submerged loggers and when there are only two.) Logger position was only significant at transect 4 and 5, which were in shallow habitat and floodplain bench habitat, respectively. The interaction between month and logger position was also statistically significant at transects 4 and 5. Transects 4 and 5 had the least reliable data, as two loggers in transect 4 had a suppressed signal indicating possible sedimentation, and loggers in transect 5 may have been in very shallow water at some time periods.

Table 9. Factors leading to statistically significant differences in mean water temperature, using two-way ANOVA (statistically insignificant factors not included)

	n	Multiple R2	Factor	df	F-ratio	p-value
Transect 1	14060	0.494	Month	5	2736.21	0.0
Transect 2	10614	0.584	Month	2	1324.55 (Jan-Mar) 537.965 (Apr-May	0.0 0.0
Transect 3	10416	0.526	Month	5	2306.80	0.0
Transect 4	12148	0.671	Month Position Interaction	5 3 15	4591.99 149.67 67.01	0.0 0.0 0.0
Transect 5	10730	0.515	Month Position Interaction	2 4 8	1413.85 (Jan-Mar) 344.18 (Apr-June 52.42 (Jan-Mar) 49.07 (Apr-Jun) 31.57 (Jan-Mar) 3.33 (Apr-Mar)	0.0 0.0 0.0 0.0 0.0 0.36
Transect 6	13144	0.599	Month	5	3925.93	0.0

This analysis implies seasonal patterns most affect water temperature, and it is difficult to make meaningful conclusions about which habitat types are most likely to display lateral heating differences. (See Appendix A for box and whisker plots of monthly water temperature by logger position for all transects.) Data from transect 6, which remained in place through summer 2007, and from the instantaneous summer transects imply that margin warming may impact instream habitat during summer months, but no clear changes could be detected during winter or spring months. Presence and quality of riparian vegetation was ignored during all lateral thermal diversity monitoring (and was largely absent). Future studies should examine the role of riparian vegetation in eliminating or minimizing summer margin heating trends.

Side Channel Longitudinal Profiles

The two side channels monitored along Nelson Ranch were narrow (< 1.5 m across) with negligible velocity. One profile was deployed in a side channel near transect 5 (Figure 42). On January 27, the channel was frozen, so a logger was set on top of the frozen channel in the hopes water temperature would be recorded when the ice melted and the side channel could provide useful fish habitat. On March 7, two additional loggers were added at the top and bottom of the channel. However, by mid-March the loggers were already exposed to air. No useable data was obtained from this profile.

Temperature loggers were placed in a second side channel northeast of the main channel near site 6 (Figure 42). During January 28, 2007 sampling, this side was mainly dry, with ice in the upstream portions. A temperature logger was deployed under the ice, in approximately 0.05m of water. Filamentous algae were growing in this side channel despite the ice, and water temperature was 3.3ºC under the layer of ice. Two additional loggers were placed in the side channel in early March when the ice had melted and more water was present. However, by early April, the loggers in the side channel were already exposed to air, indicating side channels at Nelson Ranch do not provide useable fish habitat for long periods during spring following dry years.

Spring Inflow and Tailwater Return

Nelson Ranch spring inflow and tailwater return were monitored to increase understanding of return flow conditions and recommend management actions to mitigate for warm water returns. Additional tailwater returns were monitored at Meamber Ranch, located near the DWR weir in the lower Shasta River. This section first describes flow and water temperature monitoring at Dream Spring and tailwater return at Nelson Ranch, followed by the tailwater monitoring analysis conducted at Meamber Ranch. Results show all three sources have high thermal variability and, at times, are sources of warm inflows to the Shasta River. All three sources also contribute a low volume of water to the Shasta River, creating local pockets of warm water during certain hours, but having little effect on a larger scale. The cumulative effects of tailwater returns over the length of the Shasta River were not analyzed.

At all sites, stage was measured with Global Water Instrumentation, Inc. water level logger pressure transducers (model WL15). Flow was calculated differently in all channels, based on the type of measurement device installed, and is discussed below. Pressure transducers were programmed to record water depth every hour, and are accurate to +/-0.2% of the full range of 0.914 m (0.2 cm) over the range of temperatures from 2-21°C. All hourly temperature measurements were obtained using Hobo Water Temp Pro V1 and V2 temperature loggers, manufactured by Onset Computer Corporation, accurate to +/- 0.2°C.

Dream Spring

Dream Spring emerges from a hillside, where water initially spreads over an area approximately 6m wide (Figure 45). Within 15 – 20m, spring water flows into an abandoned irrigation ditch, which runs parallel to the Shasta River for approximately 70m before joining the Shasta River. The average depth of Dream Spring in the irrigation channel is less than 15cm. In summer months, vegetation is variable, with sections of the channel shaded by nettles, greasewood, or emergent aquatic vegetation. During winter months, the channel is open to direct sunlight. This sub-section describes data collection methods, problems, and preliminary data; and proposes future options for Dream Spring, such as reducing pumping by using the spring to water cattle, or piping water directly to the Shasta River to reduce transit time and atmospheric heating.

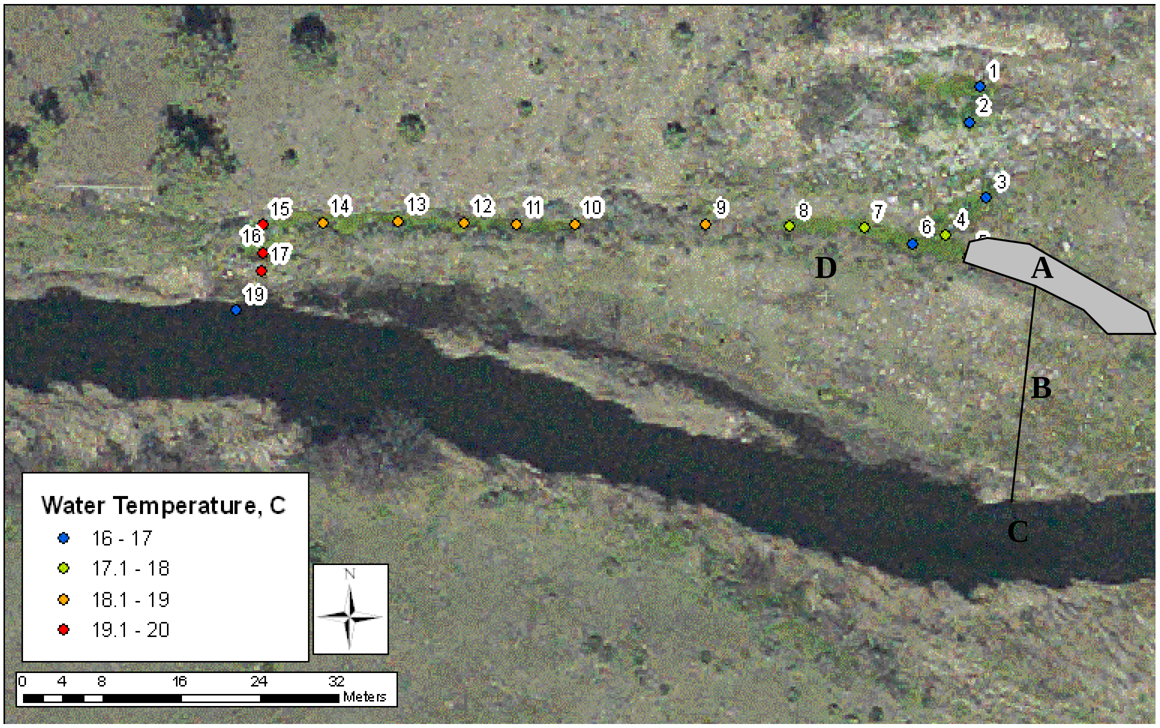


Figure 45. Dream Spring sampling locations and temperatures (sampled 8/23/06 13:00-14:00)

Flow at Dream Spring was calculated using a sharp-edged, 90° V-notch weir in the irrigation channel adjacent to the Shasta River. It was installed on 7/18/06 and removed on 4/15/07. The pressure transducer measured stage upstream of the weir to allow direct calculation of flow (Figure 45 - site 7). Discharge was calculated for a small, fully contracted 90° V-notch weir using the equation $Q = 2.49H^{2.48}$ (USBR, 2001; Aisenbrey et al., 1978), where H = head.

A temperature logger was installed about 8m upstream of the weir pond so that ponding did not affect recorded temperature (Figure 45 - site 4). The logger was installed with the sensor pointed downward in approximately 2.5cm of water. A second temperature logger was installed in November in the upper portion of Dream Spring approximately 10m downstream from where it emerges from the hillside (Figure 45 - site 3). This logger was installed for redundancy and to better understand water temperature along the Dream Spring channel, and was also positioned with the sensor facing downward in approximately 7.5cm of water. When the weir was removed in April, the upper temperature logger was moved where the spring emerges from the hillside and a third temperature logger was placed in the Dream Spring channel just before it reaches the Shasta River. All temperature loggers were protected with neoprene boots.

The irrigation channel that Dream Spring flows into has a low slope, causing water to back up in the upstream portion of the irrigation ditch for approximately 30m. (Figure 45 - area A). Flow data was compromised by likely subsurface flow between the irrigation channel and the Shasta River, which reduced weir pond stage by an unknown amount. From December 2006 to April 2007, gopher holes and subsequent leakage were additional ongoing problems.

Initial measurements suggest that the discharge in the spring varied seasonally, with a slow reduction in flow from approximately 0.065 cfs in July to approximately 0.03-0.04 cfs in August (Figure 46). However, maintenance problems outlined above detrimentally affected results. Discharge increased dramatically on October 4 and November 9, days major seeps and leaks were repaired. These data provide a lower bound for Dream Spring flow, although actual discharge from Dream Spring may be much higher. The only data error in the water temperature record occurred when the sensor on the logger above the weir became covered in sediment from August 5 to 23.

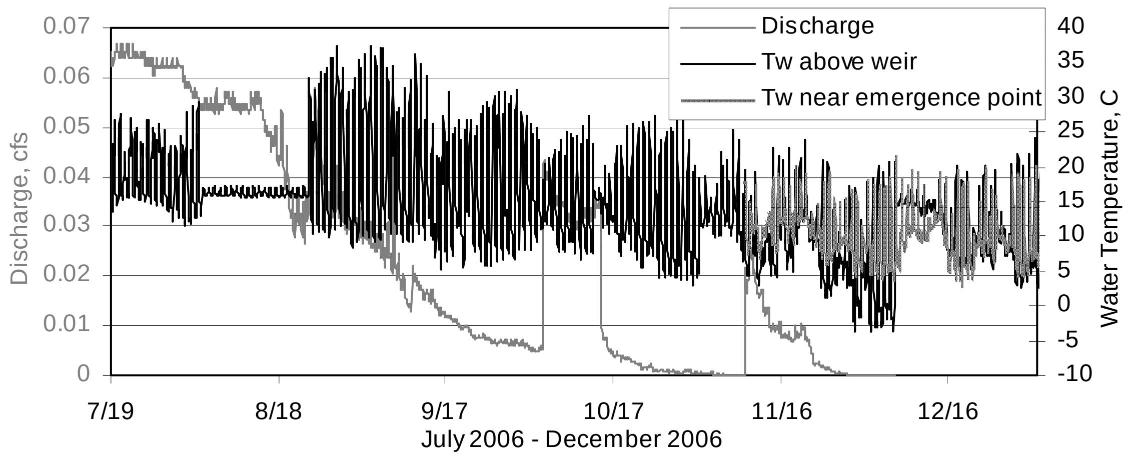


Figure 46. Dream Spring discharge and water temperature

Overall, water temperature from Dream Spring can be considerably warmer than the Shasta River (Figure 47). Interestingly, Dream Spring did not clearly show atmospheric heating between its emergence and its downstream end where it joins the Shasta River, probably because water temperature is already so warm where Dream Spring emerges (Figure 48). Dream Spring is shallow throughout its course to the Shasta River, and sunlight may have been influencing logger measurements, despite the downward orientation of their sensors. Maximum recorded temperature was 24.2°C and 29.4°C for December 2006 and January 2007, respectively; although maximum recorded air temperature at Nelson Ranch was 16°C and 16.4°C for December and January, respectively. Thermal variability in Dream Spring declined in summer, possibly due to emergent riparian vegetation providing shade.

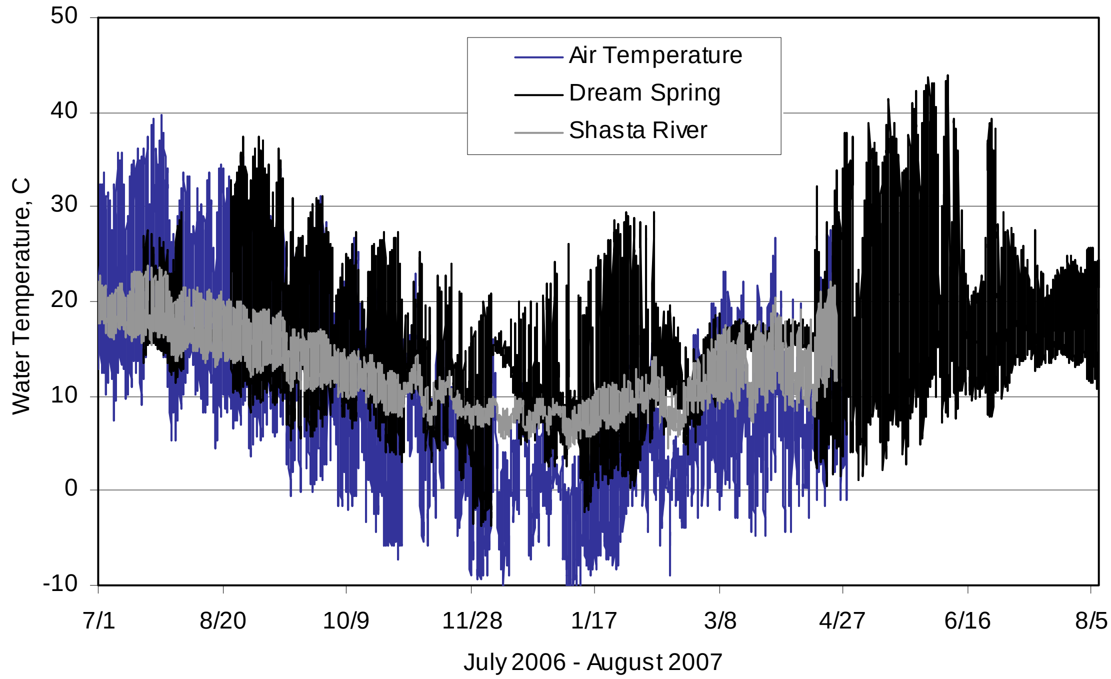


Figure 47. Air temperature and water temperature at Dream Spring and the Shasta River

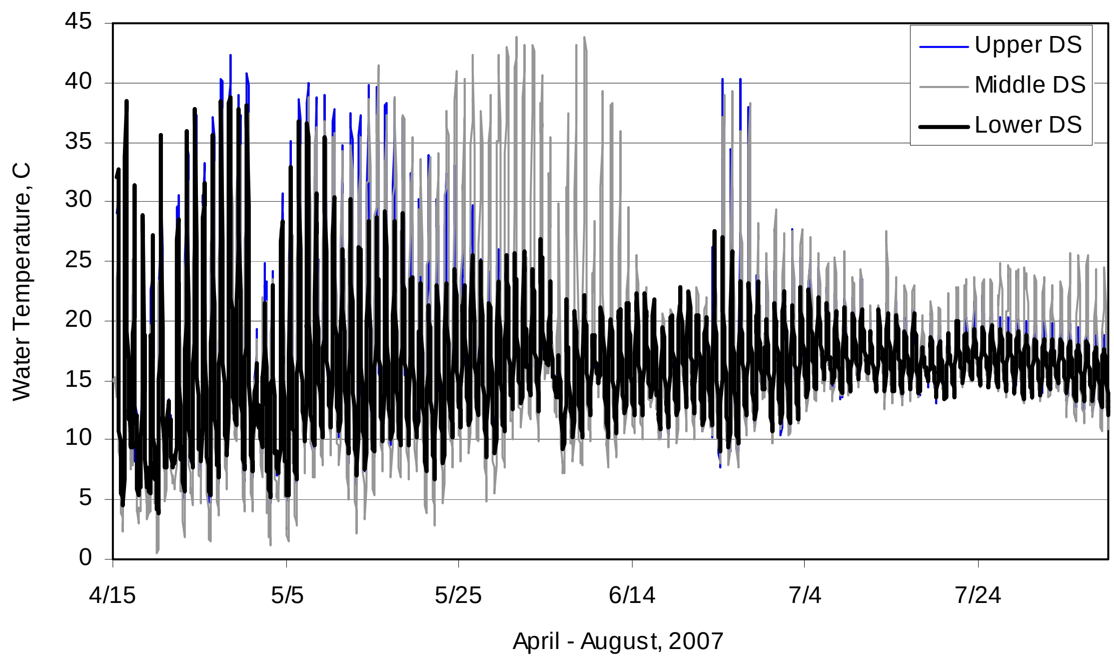


Figure 48. Water temperature at upper, middle, and lower Dream Spring

Data suggest Dream Spring is not a constant source of cool water for the Shasta River under current conditions. Although mean monthly water temperature is similar to the Shasta River, Dream Spring has high thermal variability, and is often already warm at its emergence point (Table 10, Figure 48). Water temperature has high daily variability because the channel is shallow and distance to the Shasta River is longer than necessary. The Dream Spring channel is less than 100m, and parallels the Shasta River for approximately 70m. Additional solar heating may occur prior to the emergence point if the spring is fed from a surface water source such as the pond near the ranch house. However, tracing the source of Dream Spring is outside the scope of this study.

Table 10. Minimum, mean, and maximum temperatures (°C) by month at Dream Spring (near the weir), Shasta River at the upstream boundary of Nelson Ranch, and Nelson Ranch return flow

	Dream Spring			Shasta River			Return Flow		
	Minimum	Mean	Maximum	Minimum	Mean	Maximum	Minimum	Mean	Maximum
July-06	12.2	18.8	27.5	14.3	19.1	23.8	9.4	22.3	46.4
August-06	8.3	17.8	37.3	12.1	17.1	21.5	7.2	20.0	43.4
September-06	5.3	18.0	37.5	10.6	14.6	19.2	9.4	16.4	23.9
October-06	3.0	14.0	27.4	7.8	11.5	16.0	-5.1	10.5	31.8
November-06	-3.5	10.2	25.2	6.7	9.7	14.4	-4.4	8.6	29.4
December-06	-3.8	9.8	24.2	5.5	7.7	9.9	-1.5	6.3	19.7
January-07	-2.3	8.0	29.4	4.5	7.9	12.2	1.2	5.1	12.9
February-07	0.6	10.7	29.4	4.4	9.4	14.1	0.3	6.3	13.2
March-07	8.0	15.0	18.8	7.2	12.4	20.5	-0.4	9.4	24.5
April-07	0.6	15.5	37.8	7.8	13.2	22.2	-2.0	11.3	31.6
May-07	1.2	18.9	43.8				7.1	14.4	25.9
June-07	7.2	17.6	43.9				10.4	17.1	26.4
July-07	10.5	17.7	27.7				13.9	18.8	28.2

Dream Spring is shallow with a low flow volumes, and greater thermal variability than the mainstem Shasta River. This small inflow cools faster than the Shasta River, possibly providing localized thermal refuge at its confluence during some times of day. This research indicates that under optimal conditions, Dream Spring may provide only minor habitat improvement. Under its current configuration, with maximum summer temperatures near 40°C, Dream Spring can be detrimental to instream habitat conditions. Two future options for Dream Spring have been identified. First, water from Dream Spring can be pooled and used for watering cattle. This option is especially promising if it allows a reduction in the Nelson Ranch diversion from the Shasta River.

A second option is to experiment with maintaining initial Dream Spring temperature by piping the spring directly from its emergence point to the Shasta River. This assumes that atmospheric heating does sometimes occur. A direct benefit of this approach would be reduced seepage loss, resulting in more accurate discharge measurement. When spring water emerges at temperatures so warm that atmospheric heating does not occur, this alternative would have no benefit. Topography features, such as a steep embankment and the abandoned irrigation channel, are challenges.

Nelson Ranch Tailwater Return

Water is diverted from the Shasta River at an unmetered pumping station near the upstream property boundary of Nelson Ranch. Tailwater returns to the Shasta River approximately 500m downstream via a return channel that drains fields on the southern portion of Nelson Ranch. Water depth in the return channel averages approximately 10cm. Herbaceous riparian vegetation lines the channel during spring and summer, and in the winter it is open to solar radiation. The clarity of the water varies and sometimes has a noticeable reddish or brownish hue.

The volume and temperature of tailwater at Nelson Ranch was monitored to increase understanding of its thermal effects on the Shasta River, and to highlight promising alternatives to improve the efficiency of the ranch and fish habitat in the Shasta River. This analysis only examines tailwater that returns to the Shasta River via the return ditch, ignoring possible subsurface or overland flows.

A three-inch Parshall Flume was installed in the tailwater channel in late July 2006 by the UC Davis Watershed Sciences Center. A pressure transducer was deployed with the flume to measure stage. The UC Davis Watershed Sciences Center oversaw maintenance of the Parshall Flume, downloaded stage data, and calculated flow based on recorded data. A temperature logger was deployed in the tailwater channel approximately 10m upstream of the flume.

Baseflow of approximately 0.14 cfs is maintained in the return flow channel throughout the year (Figure 49). During irrigation season, flow pulses reached 1.6 cfs. Pulses occasionally occur outside of irrigation season from local storm runoff. Mean return flow water temperature is 3-4°C warmer than mean Shasta River temperature during summer, and is 1.5-3°C cooler during winter. Like Dream Spring, tailwater temperature is much more variable than the Shasta River (Table 10, Figure 50). Tailwater temperature exceeded 40°C in July and August 2006.

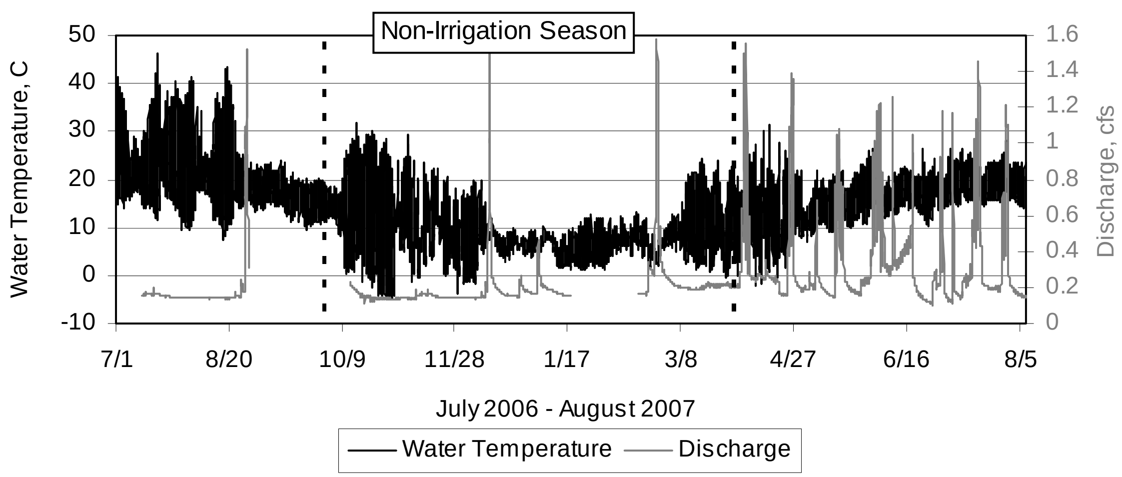


Figure 49. Nelson Ranch tailwater return and water temperature

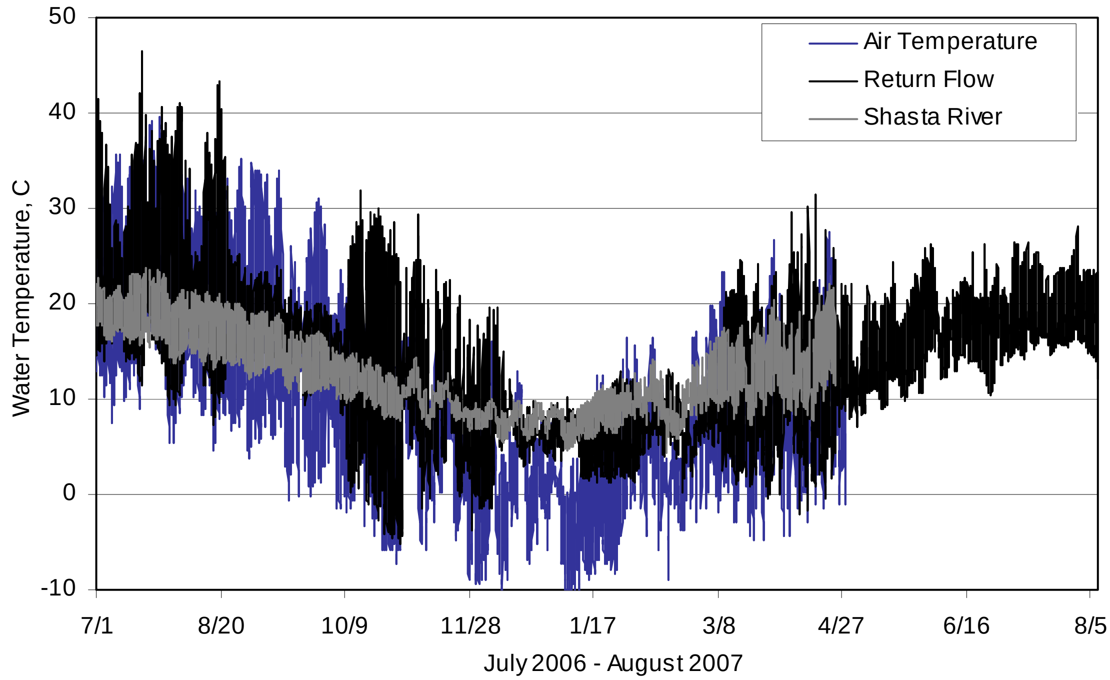


Figure 50. Air temperature and water temperature at Dream Spring and the Shasta River

Return flow volume is negligible compared to that of the Shasta River (Figure 51). Regardless, tailwater temperature can be much higher than Shasta River water temperature, creating a local influx of warm water during some times of day (Figure 38). This primarily occurs during summer, when water temperature in the Shasta River is often already near critical limits for salmon species.

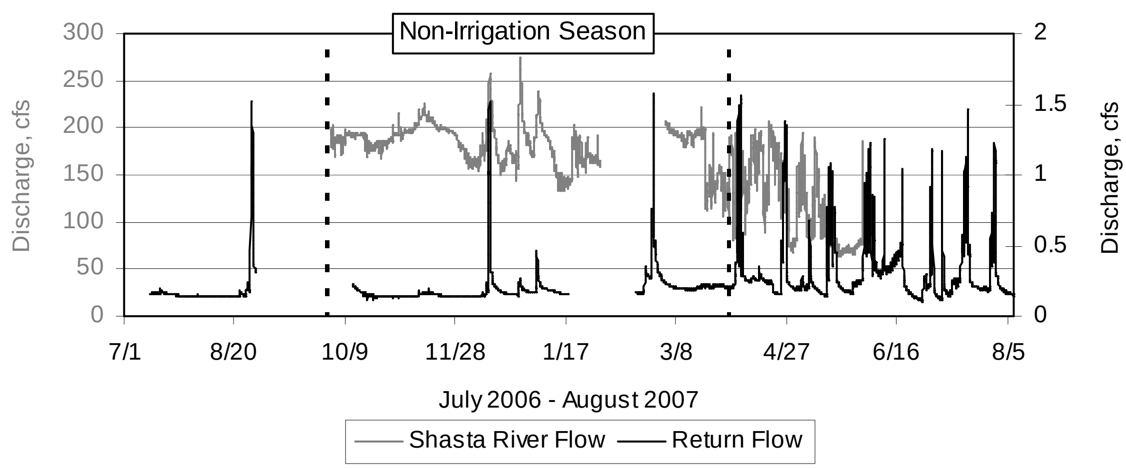


Figure 51. Discharge in the Shasta River at the upstream boundary of Nelson Ranch and the return flow channel

Overall channelized return flow to the Shasta River from Nelson Ranch averages 0.14 cfs, and peaks to 1.6 cfs. These results would be most beneficial if pumping from the Shasta River was measured. This would give important information regarding efficiency ratios. In depth analysis of tailwater return, including possible subsurface or overland flow would help to more correctly quantify tailwater return to the Shasta River.

Meamber Ranch Tailwater Return

The Meamber property abuts the Shasta River for approximately ¼ mile. A weir operated by DWR monitors stage, flow, and water temperature in this section of river. Return flow discharge and water temperature were monitored at Meamber Ranch from 9/7/2006 - 10/5/2006. Tailwater at this site flows from fields to a vertical culvert, which acts as a catchment basin (Figure 52). Water enters the vertical culvert through a grated 34.6 cm circular orifice. From the vertical culvert, water returns to the Shasta River via a 20.3 cm underground pipe.

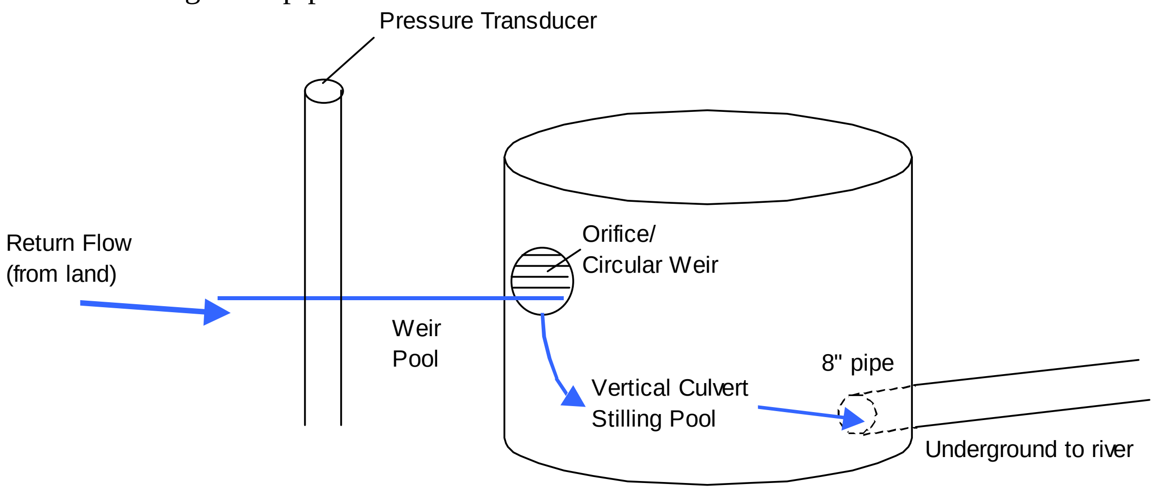


Figure 52. Schematic of Meamber Ranch agricultural return flow

A pressure transducer was installed on 9/7/06 to measure stage upstream of the vertical culvert that drained water to the Shasta River. It was suspended in a 1.5 m PVC tube so that the sensor was approximately 10 cm from the bed of the channel, and was removed on October 5, at the end of the irrigation season. When stage was below the top of the opening, the opening acted as a circular weir and discharge (Q) was calculated using equations 40 and 41 (Erickson et al., 2007), where a weir coefficient (C_d) is a function of relative head (H) with respect to weir diameter (D). Head was never greater than the top of the circular opening.

$$C_d = 0.555 + \frac{1}{110\left(\frac{H}{D}\right)} + 0.041\left(\frac{H}{D}\right) \quad \textbf{(40)}$$

$$Q = 0.0039C_d\left[10.12\left(\frac{H}{D}\right)^{1.975} - 2.66\left(\frac{H}{D}\right)^{3.78}\right](10D)^{\frac{5}{2}} \quad \textbf{(41)}$$

Two temperature loggers were deployed on 7/17/06. One logger was connected to a stake in the return flow channel, and was placed approximately 15 cm above the bed (approximately 13cm below the surface when flow through the return ditch was negligible). The other logger was tied to the staff gage at the USGS weir in the Shasta River, approximately 40cm below the water surface. Both loggers were protected with neoprene boots and were installed with their sensors facing downward. Shasta River discharge at the DWR weir was downloaded from CDEC (2007).

Between September 7 and October 5, agricultural return flow ranged from zero to 3.97 cfs, and averaged 0.4 cfs (Figure 53). During this period, water temperature ranged from 2.8^{o}C to 33.3^{o}C, and averaged 13.3^{o}C.

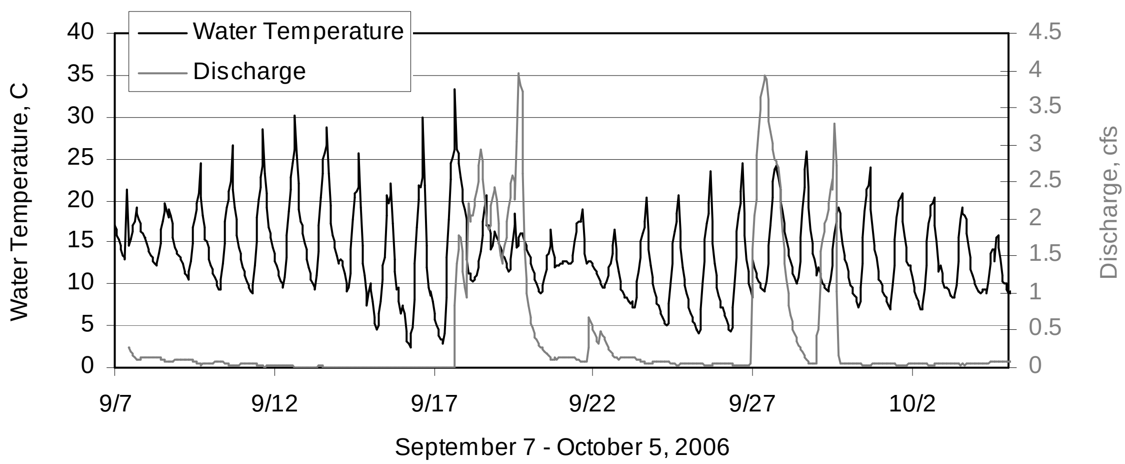


Figure 53. Meamber Ranch agricultural return flow and water temperature

Agricultural return water temperature was monitored from July 19 to October 5. Over this longer period, average water temperature was 17.2^{o}C, and ranged from 2.8^{o}C to 53.3^{o}C (Figure 54). Regression analysis shows that water temperature trended downward by over 11.1^{o}C from mid-summer to early-fall. Table 11 lists basic statistics for water temperature by month.

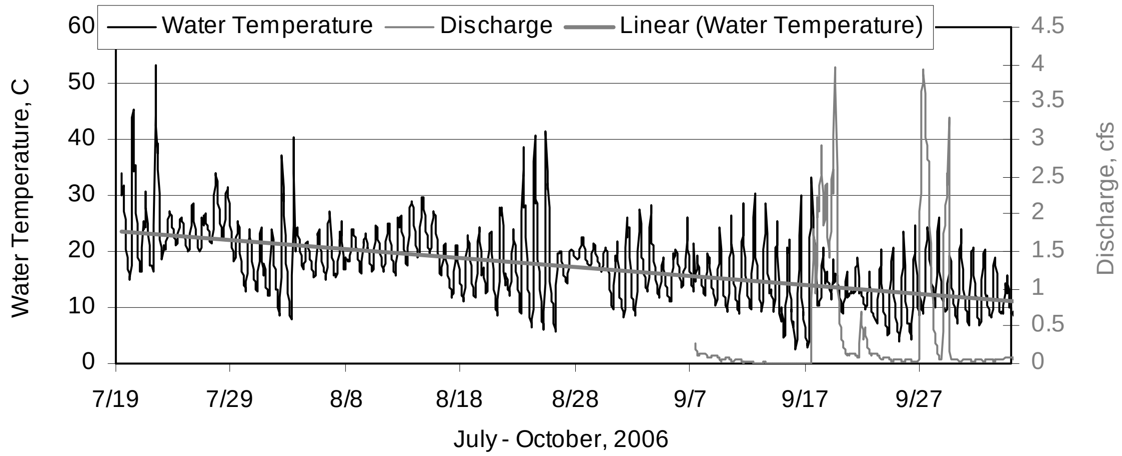


Figure 54. Meamber Ranch agricultural return flow, water temperature, and temperature linear regression

Table 11. Minimum, mean, and maximum temperatures (°F) by month

	Return Flow			Shasta River		
	Minimum	Mean	Maximum	Minimum	Mean	Maximum
July	12.9	23.5	52.8	17.4	22.5	25.6
August	5.8	18.8	41.6	15.1	19.6	23.1
September	2.5	14.3	33.4	11.8	16.2	20.2
October	6.9	11.9	20.9	12.1	14.0	16.4

From July 17 to October 5, average water temperature of the Shasta River at the DWR weir was 18.3°C, and ranged between 11.8°C and 25.6°C (Figure 55, Table 11). Discharge averaged 91 cfs, and ranged between 39 - 219 cfs. Regression analysis shows that the water temperature of the Shasta River decreased by approximately 8.9°C from mid-summer to early-fall, and discharge increased by approximately 45 cfs.

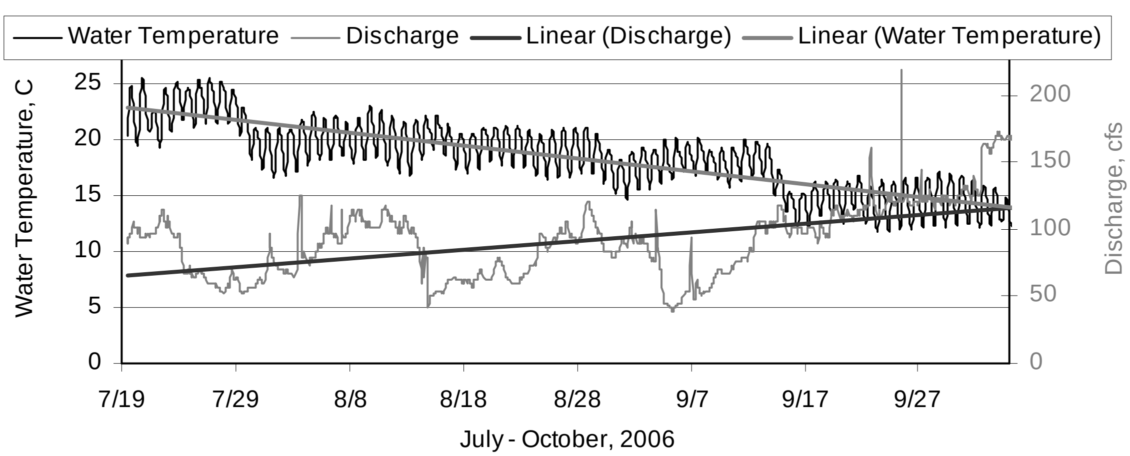


Figure 55. Shasta River flow, water temperature, and linear regressions

Overall, agricultural return flow to the Shasta River from the Meamber property remained low during the month that was monitored. Average water temperature was comparable between the agricultural return flow channel and the Shasta River. However, agricultural returns had greater thermal variability, resulting in a net cold water source at night and a net warm water source during the day. For example, during a single day (September 17), the water temperature range (maximum minus minimum) of return flows exceeded 27.8^{o}C. When return flows increased from negligible to the order of 2 cfs, as exhibited September 18-20, the diurnal range in return flow temperatures decreased, but was still 5.6^{o}C. Ongoing monitoring throughout the bulk of irrigation season will increase knowledge about return flow quantity and timing.

Dream Spring and Tailwater Return Conclusions

Water temperature at Dream Spring and both tailwater return sites have very high thermal variability. Dream Spring and tailwater are a net warm water source to the Shasta River in afternoons during summer. Flow volume of all of these sources is relatively small. At Nelson Ranch, thermistors located upstream and downstream of these known warm water sources did not show obvious heating trends, as flow volume is too small to change instream thermal conditions. As exploratory temperature probing and instantaneous transects near the Nelson Ranch tailwater return showed above, the warm water inflow is substantial enough to create local pockets of warm water during summer, but not enough to make noticeable changes to the thermal conditions of the river, as a whole. Managing tailwater and other potential warm water inflow throughout the Shasta River is a challenge because it is difficult to quantify small, individual effects of these warm water sources, but collectively throughout the Shasta River, warm water inflows may detrimentally affect thermal conditions and instream habitat for cold-water fish species. Ongoing studies are recommended to increase understanding of potential return flow conditions and support future water management actions.

Discussion

The temperature observations described here identify several key insights into small-scale thermal conditions on the Nelson Ranch.

- Longitudinal thermal diversity exists, primarily in response to upstream conditions (both natural and anthropogenic) and meteorological influences on the river as it travels downstream.
    - Maximum water temperature near the downstream property boundary of Nelson Ranch occurs at night.
    - Data suggest a volume of warm water is being inherited upstream of Nelson Ranch.

- Lateral variability is apparent near river margins, and is most pronounced during summer.
    - Riparian vegetation, both herbaceous and woody, may provide benefits for such margin habitat.

- During summer, small, localized cool water refugia associated with subsurface flow, seeps, and/or springs were identified on the Nelson Ranch with temperatures up to 1-2°C cooler than mainstem river conditions.
    - Cool water refugia and return flows were generally small in size and/or magnitude and did not appear to have an appreciable influence on overall mainstem temperatures.

- Winter mainstem temperatures were largely uniform, laterally.

- Side channels were typically frozen during winter months, and became dry in early spring of dry water years.

- Additional observations of potential refugia, springs, and return flows would increase information and improve understanding of thermal variability; as well as impacts on anadromous fish production and appropriate management strategies.

- Monitoring illustrates longitudinal and lateral variability that is not apparent with other study approaches (such as current modeling efforts).
    - It is important to continue field monitoring to understand small-scale variability.

The field studies outlined in this chapter provide detail regarding the thermal conditions of the Shasta River. The modeling studies in the following two chapters help increase understanding of year-round flow and temperature conditions in the Shasta River, and highlight the most promising management alternatives to enhance cold-water habitat while considering water use efficiency. Yet, modeling results described in the following chapters cannot have the level of detail described here. The small-scale results of thermal diversity studies will be used to interpret model results, when detailed thermal conditions of the Shasta River cannot be simulated.

Finally, the thermal conditions of Nelson Ranch should be considered in conjunction with other habitat data. When paired with hydrology and geomorphology data, as well as known locations of coho, Chinook, and steelhead, this data will help to make a complete picture of instream habitat conditions of Nelson Ranch, and to create a better understanding of habitat potential for the Shasta River. Access to the Shasta River via Nelson Ranch and Meamber Ranch has contributed to increasing scientific knowledge of the thermal conditions of the Shasta River. Access at more sites would increase existing knowledge and aid management of the Shasta River.

References

Aisenbrey, Jr., A.J., R.B. Hayes, J.J. Warren, D.L. Winsett, R.B. Young. U.S.B.R. 1978. Design of Small Canal Structures. U.S. Dept. of the Interior. U.S. Government Printing Office. Denver, CO.

CDEC (CA Dept. of Water Resources – California Data Exchange Center). Available online (http://cdec.water.ca.gov/). Accessed 4/2007.

Erickson, A., J. Gulliver, R. Hozalski, O.Hohseni, J. Nieber, B.N. Wilson, P. Weiss. 2007. Water budget measurement. In Assessment of Stormwater Best Management Practices, ed. J.S.

Gulliver and J.L. Anderson. St. Paul, MN. University of Minnesota. Available at: http://wrc.umn.edu/outreach/stormwater/bmpassessment/guidancedocreview/pdfs/chapter4-water-budget-measurement.pdf. Accessed 3/2007.

Jeffres, C., E. Buckland, J. Kiernan, A. King, A. Nichols, S. Null, J. Mount, P. Moyle, M. Deas,. 2008. Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California – Water Year 2007. Prepared for the California Nature Conservancy by U.C Davis Center for Watershed Sciences and Watercourse Engineering, Inc.

NRC (National Resources Council). 2004. Endangered and Threatened Fishes in the Klamath River Basin: Causes of Decline and Strategies for Recovery. Committee on Endangered and Threatened Fishes in the Klamath River Basin, National Research Council. The National Academy Press.

The Nature Conservancy website. http://www.nature.org/wherewework/northamerica/states/california/features/nelson.html. Accessed 8/07.

USBR (U.S Bureau of Reclamation). 2001. Water Measurement Manual: A guide to effective water measurement practices for better water management. U.S. Department of the Interior. U.S. Government Printing Office. Washington, DC 20402. Available at http://www.usbr.gov/pmts/hydraulics_lab/pubs/wmm/. Accessed 6/2006.

Chapter 5: Year 2001 Shasta River Model Simulations for Flow and Water Temperature

This chapter describes the data and assumptions for the Shasta River model simulating unimpaired and current conditions for the year 2001. The Tennessee Valley Authority's River Modeling System (TVA-RMS v.4) is used to simulate flow and water temperature. Other water quality conditions are not examined. RMS is described, with emphasis on governing equations and necessary input data. RMS has previously been applied to the Shasta River to evaluate methods to reduce instream water temperature and for the California's Regional Water Quality Control Board (RWQCB) (Abbott, 2002; Deas et al., 2003; Geisler, 2005). A brief description of previous applications of the RMS model to the Shasta River is included. Model inputs including geometry, meteorology, coefficients, tributary boundary conditions, and initial flow and water temperature conditions are described and assumptions explained.

This chapter concludes with analysis of eight model runs: current conditions, Shasta River minimum instream flows, reducing or relocating the GID diversion, Nelson Ranch return flow analysis, restoring riparian vegetation, fully restoring Big Springs, removing Dwinnell Dam, and unimpaired conditions. Although there is uncertainty in estimates and approximations, this analysis largely constrains the problem to provide a reasonable estimate of current and potential flows and temperatures for a representative year in the Shasta Basin, and demonstrates that additional data collection is needed to improve estimates.

Model description

RMS is used to simulate flow and water temperature in California's Shasta River because it has previously been applied to the Shasta River, has shading logic, is open source, and is supported by TVA. RMS is a 1-dimensional (longitudinally), physically based numerical model composed of a hydrodynamics module (ADYN) and a water quality module (RQUAL), both with Fortran source codes (Hauser and Schohl, 2002). An additional fish bio-energetics component (FISH) is available, but is not used here. In this application the time step is one hour and the spatial scale is variable to accommodate the sinuosity of the stream.

ADYN (Hydrodynamics Module)

ADYN simulates dynamic tributaries at channel junctions, multiple tributaries with different boundary conditions, and distributed or point lateral inflows (Hauser and Schohl, 2002). The Shasta River is modeled as one continuous reach with tributaries as point inflows and distributed accretions and depletions. ADYN solves for water depth and velocity using one-dimensional equations for conservation of mass and momentum (St. Venant equations for unsteady flow), using a four-point implicit finite difference scheme with weighted spatial derivatives (Hauser and Schohl, 2002). The governing equations are one-dimensional equations for conservation of mass (eq. 42) and momentum (eq. 43). Secondary equations used in the conservation of momentum equation are energy slope (eq. 44), and channel contraction and expansion (eq. 45).

$$\frac{\partial Q}{\partial x}+\frac{\partial A}{\partial t}-q=0 \qquad \textbf{(42)}$$

$$\frac{\partial Q}{\partial t}+\frac{\partial (Q^2/A)}{\partial x}+gA(\frac{\partial H}{\partial x}+S_f+S_{ce})-qV_x=0 \qquad \textbf{(43)}$$

$$S_f=\frac{n^2Q|Q|}{221A^2R^{4/3}} \qquad \textbf{(44)}$$

$$S_{ce}=k_{ce}\frac{\partial}{\partial x}(\frac{V^2}{2g}) \qquad \textbf{(45)}$$

where Q is volumetric flow rate (cfs), A is cross sectional area (ft), H is water surface elevation (ft), x is distance along channel (ft), t is time (s), g is acceleration due to gravity (ft/s), V_x is x component of velocity of lateral inflow (ft/s) (zero assumed except for dynamic junctions), q is lateral inflow rate (cfs), S_f is energy slope, S_{ce} is channel contractions / expansions, n is Manning resistance, R is hydraulic radius (ft), k_{ce} is contraction / expansion loss coefficient, and V is average section velocity (ft/s).

The input to run ADYN includes channel geometry (channel cross sections, elevations, and bed slope), roughness coefficients, upstream inflow, lateral inflows, diversions, and upstream and downstream boundary conditions (Figure 56).

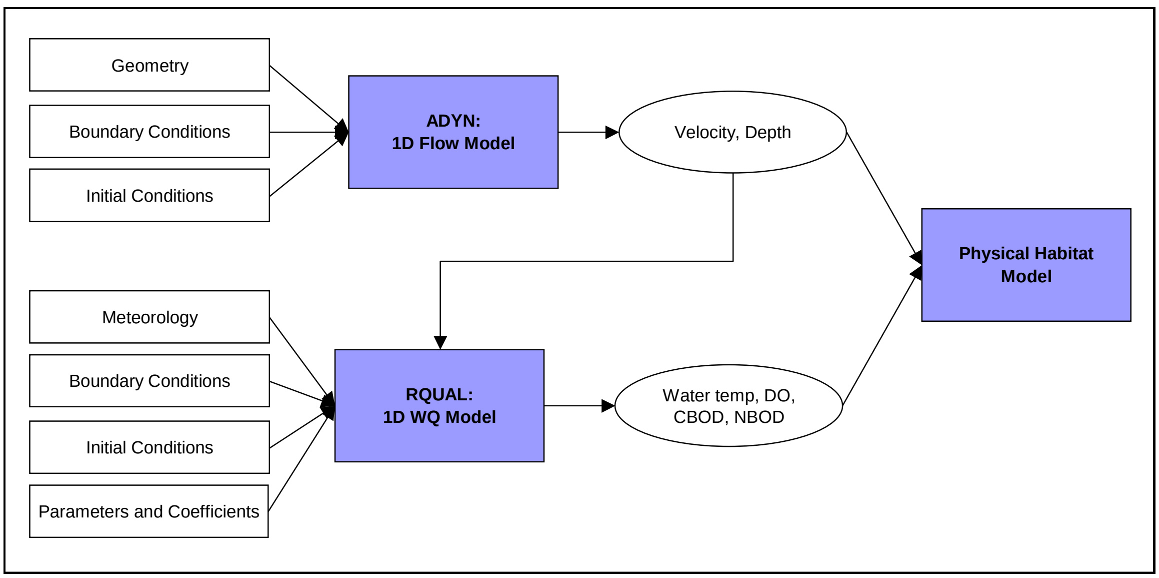


Figure 56. ADYN and RQUAL flow chart

RQUAL (Water Quality Module)

ADYN and RQUAL are run in sequence; after the hydrodynamic module has successfully run, velocities and water depths are passed to RQUAL, the water quality module (Figure 56). RQUAL solves the mass transport (advection/diffusion) equation using a Holly-Preissman numerical scheme. This module simulates the fate and transport of heat energy and constituent concentrations to represent water temperature (Tw), dissolved oxygen (DO), and carbonaceous and nitrogenous biological oxygen demand (CBOD and NBOD) (Hauser and Schohl, 2002). This model does not explicitly assess the fate and transport of nutrients or nutrient byproducts.

Water temperature is modeled using a physically based heat budget approach, which simulates the net exchange of heat at the air-water interface and the bed-water interface under specified meteorological and riparian shading conditions. Dispersion and topographic shading are ignored in RQUAL. Ignoring dispersion implies that this model is meant for systems where transport is the main mixing influence for heat and other constituents, occurring in all but very slow velocity systems. Topographic shading does not greatly influence water temperature in the Shasta River, and may only be pertinent in the lower canyon reach of the river.

RQUAL solves for mass transport (eq. 46) using equations for heat budget (eq. 47), dissolved oxygen, carbonaceous BOD and nitrogenous BOD. DO, CBOD and NBOD are ignored for this application, so those governing equations are not included here, but are found in Hauser and Schohl (2002).

$$\frac{\partial C}{\partial t} + V\frac{\partial C}{\partial x} + \sum\left[\frac{q}{A}(C - C_1) - \frac{0.186W_1}{A\Delta x}\right] - \sum\left[\frac{\partial C}{\partial t^s}\right] = 0 \qquad \textbf{(46)}$$

where C is constituent concentration (quantity/L), C_1 is constituent concentration of lateral inflow (quantity/L), W_l is wasteloading of constituent C (quantity/day), and $\sum\left[\frac{\partial C}{\partial t^s}\right]$ is the sum of internal sources and sinks of constituent C (quantity/L*s).

$$\sum\left[\frac{\partial T}{\partial t^s}\right] = (Q_{ns} + Q_{na} + Q_{bed} - Q_b - Q_e - Q_c)/D \qquad \textbf{(47)}$$

where D is mean depth (m), T is thermal energy (kcal/m^3), Q_{ns} is net solar radiation at the water surface adjusted for shading and fog (kcal/m^2*s), Q_{na} is net atmospheric (long wave) radiation (kcal/m^2*s), Q_{bed} is net heat transfer to water from the channel to the bed (kcal/m^2*s), Q_b is back radiation from the water body (kcal/m^2*s), Q_e is evaporative heat loss (kcal/m^2*s), and Q_c is conductive heat transfer (kcal/m^2*s). Deas et al. (2003) provide a detailed description of these heat budget terms.

Model input to RQUAL includes hydrodynamic data (the output from ADYN), meteorological data (air temperature, dew point temperature, wind speed, cloud cover, barometric pressure, and solar radiation), initial water quality, lateral inflow water quality, and shading from riparian vegetation (Figure 56).

Application to California's Shasta River

RMS has been used to model the Shasta River for two earlier studies. The first evaluated the potential of riparian shading and alternative flow management to reduce instream water temperature by simulating the Shasta River from four miles below Dwinnell Dam to the mouth (Abbott, 2002; Deas et al., 2003). Water temperature reduction was the primary objective of that study, and three 6-day periods were modeled in July, August, and September of 2001. Modeled alternatives included: increased instream flow, pulse flow, distributed or point source return flow management, and reach by reach shading. For that project, the RMS code was modified to more accurately represent location, height, and shade providing characteristics of spatially diverse riparian vegetation. A new sub-routine was written to allow solar transmittance and vegetation height to vary longitudinally down the river and between the left and right banks (Abbott, 2002). Those code modifications were used for this study.

Model calibration and testing was first completed for the RMS Shasta River model by Deas et al. (2003). Sensitivity analysis was conducted for model parameters, such as Manning's n, contraction and expansion coefficients, wind coefficients, and the weighting factor for spatial derivatives. Simulations were completed with ADYN with various steady-state flows ranging from 2-200 cfs to evaluate model performance. Similarly, water temperature response to flow, tree height, and transmittance changes was completed with RQUAL. The Shasta River model was calibrated for August 17-23 by comparing modeled output with measured data, and was tested by comparing July 21-27 output with measured data.

The Shasta River RMS model was again used for a flow and water temperature study prepared for the North Coast Regional Water Quality Control Board (NCRWQCB) and the UC Davis Information Center for the Environment (Geisler, 2005). For the second application, the model was extended upstream to Dwinnell Dam and river geometry was updated using 1:24K hydrography created by David Lamphear of Humboldt State University, discussed further below. The model was run for three week-long periods beginning 7/2, 8/29, and 9/17/2002, with the September period used for model calibration and the other two periods used for model testing. Sensitivity analysis was performed on sensitivity of flow to the Manning roughness coefficient, evaporative heat flux values, SOD values, and the CBOD and NBOD decay rates, as well as algal photosynthetic and respiratory rates. Geisler (2005) contains a table of all parameters values, descriptions, and references. Model parameters and coefficients from that study are left unchanged here, unless otherwise noted.

Geometry

The Lamphear hydrography has a more detailed portrayal of the Shasta River, particularly in the meandering reaches of the river between Hwy A-12 and the DWR weir. The Shasta River was represented with 999 nodes (the maximum allowed in RMS) from Dwinnell Dam to the confluence with the Klamath River, a modeled length of 40.62 miles. Nodes are not evenly spaced, meandering reaches have more nodes than straighter reaches. Overall, RMS physical representation of the Shasta River is quite accurate. Figure 57 shows RMS river points and nodes overlain on a 9 in. resolution aerial photo of The Nature Conservancy's Nelson Ranch (TNC, 2006). Each RMS node has accompanying cross-sectional geometry data in which the shape of the channel is described with five points (Figure 58). Geometry data includes distance from the first point of the cross section, and associated elevation. The river geometry developed by Geisler (2005) is unchanged for this study. Full methodology is presented in Geisler (2005) and Abbott (2002).

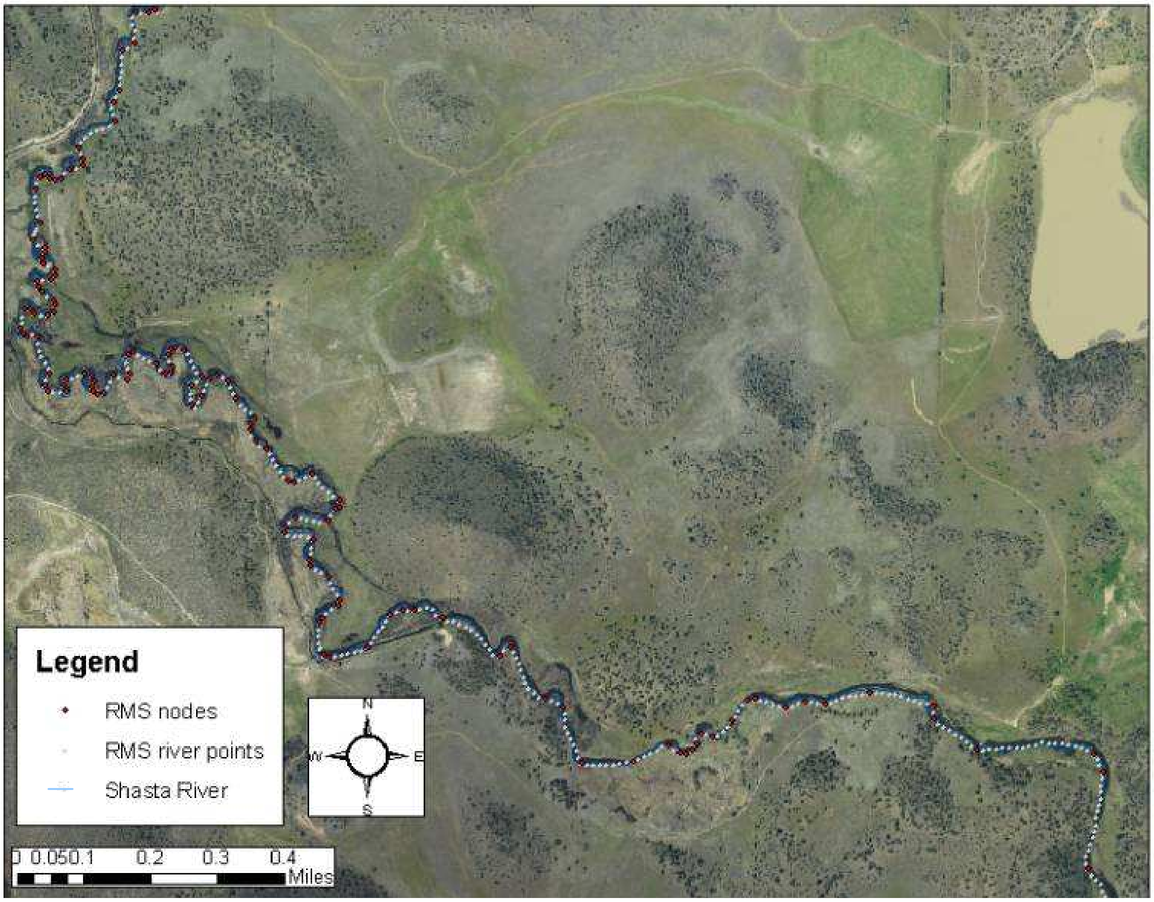


Figure 57. RMS Shasta River depiction and nodes at Nelson Ranch

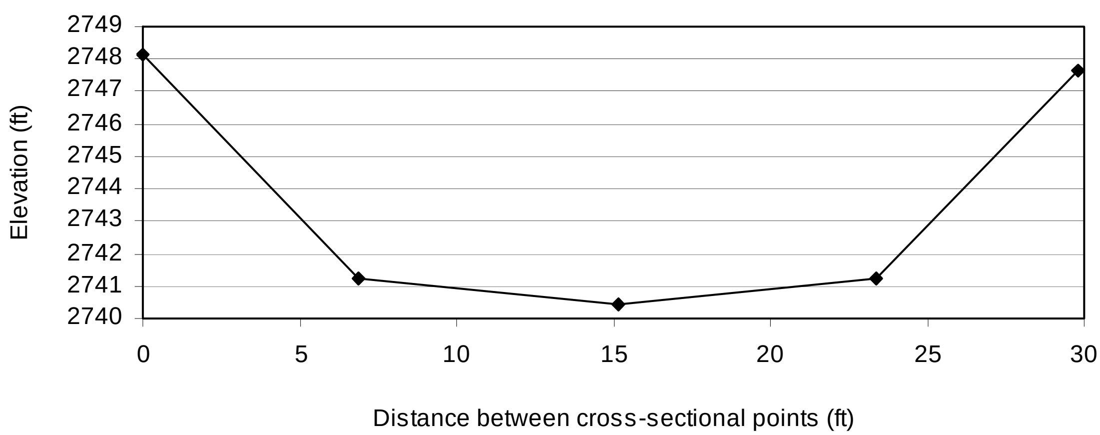


Figure 58. Representative RMS river cross-section

Unimpaired Conditions

Pre-development conditions represent an estimate of the historic thermal regime of the Shasta River prior to groundwater pumping, construction of Dwinnell Dam, stream impoundments, diversions, and land use changes. Hydrology and water temperature input data for unimpaired conditions are discussed in the following paragraphs.

Hydrology

Unimpaired monthly hydrology estimates were required for the Shasta River at Dwinnell Dam, Parks Creek, Big Springs, the Little Shasta River, and Yreka Creek. Unimpaired inflow for the summer period (May –September) for Dwinnell Dam was derived from DWR Watermaster Service records (as outlined in Deas et al., 2004). DWR Watermaster service records from 1950-55 were used for the remaining months of the year because year-round data were available for this period (typically Watermaster service records only include irrigation season observations). Average monthly flow data for Parks Creek and the Little Shasta River from May to September was from Shasta River unimpaired flows (CDWR Watermaster, 1930-1990; Deas et al, 2004). For the remaining months, flow for Parks Creek and the Little Shasta River was determined by water balance $((Q_{mouth} - \Sigma Q_{Dwinnell, Big Springs, Yreka, Depletion})/2)$ for each tributary. Big Springs records were derived from the Department of Public Works, Division of Water Rights water supply report (DPW-DWR, 1925). Flow for Yreka Creek was calculated by watershed area based on communication with the North Coast Regional Water Quality Control Board. The basin area of Yreka Creek is 6.64% of the total area, so 6.64% of total flow at the mouth of the Shasta River is applied to Yreka Creek each month.

A seasonal depletion was included to balance the monthly flows at the mouth based on the DWR unimpaired flow study (CDWR, 1998). The Shasta River was assumed to lose water to groundwater, evaporation, and evapotranspiration of riparian vegetation from Dwinnell Dam to Yreka Creek (river mile 40.62 – 7.79). Losses were estimated to be 20% of the flow at the mouth from May to September, and 10% in the remaining months. These values are consistent with typical field losses (FAO, 1989). During late spring and late summer periods the water balance did not close. Additional seasonal variation in depletions was considered for these months, but the results did not appear reasonable. Thus, this discrepancy remains in the model simulations. The resulting net error in annual runoff is approximately 0.1 percent – the largest discrepancies occurred in May and September, with approximately 18% over-estimation and 26% under-estimation of flow at the mouth. Considering the total unimpaired flow quantity in the river (always greater than 143 cfs), the timing of overestimation (late spring), and underestimation (late summer/early fall), the estimates appear reasonable and are conservative for the fall. The flow data are summarized in Table 12. For all boundary inflows, daily data was linearly interpolated from monthly averages assigned to the middle of each month (Figure 59). This approach averages winter peak flows so there is little variability and possibly higher winter base flow.

Table 12. Monthly average boundary flow

Date	Dwinnell	Parks	Big Springs	Little Shasta	Yreka	Depletion*	Mouth DWR Unimpaired Flow Study	Mouth Water Balance	Difference
	cfs	cfs	cfs	cfs	cfs	cfs	cfs	cfs	cfs
1/15/2000	127	112	117	112	30	45	454	453	0
2/15/2000	177	81	114	80	29	44	437	437	0
3/15/2000	102	110	111	109	28	42	417	417	0
4/15/2000	105	52	107	51	20	30	304	305	0
5/15/2000	96	71	104	49	16	49	244	287	-43
6/15/2000	65	40	107	30	14	44	218	212	5
7/15/2000	38	13	111	17	10	31	155	158	-3
8/15/2000	32	7	114	14	10	31	153	147	6
9/15/2000	31	6	117	13	13	39	194	143	51
10/15/2000	21	70	121	69	18	27	272	272	0
11/15/2000	43	95	124	94	23	34	344	345	0
12/15/2000	122	88	121	88	27	41	405	405	0

* Losses to groundwater, evaporation, and riparian vegetation evapotranspiration

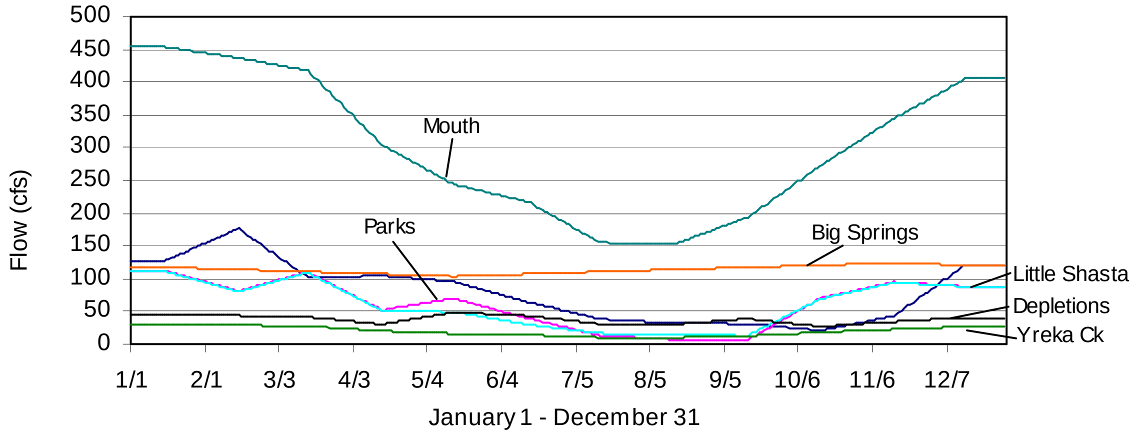


Figure 59. Unimpaired daily flow timeseries interpolated from monthly average flow records

Initial Downstream Boundary Condition

In previous RMS Shasta River models, a Manning equation rating was used with a dynamic approximation of the energy slope to determine downstream flow and stage. However, with large flows this boundary condition proved challenging to produce a viable initial condition for the model. To overcome this condition, a downstream elevation boundary condition was applied, so that the downstream boundary condition was fixed at 2,042 ft (622 m) above sea level – an elevation that is typical of mean annual flows near the mouth (node 999 – river mile 0). The imposition of this stage boundary condition did not significantly affect model results. Model output from node 994 (approximately 0.9 miles upstream from the mouth) was used for conditions at the mouth to avoid any effects from the boundary condition.

Water Temperature Boundary Conditions

Mainstem, Parks, Little Shasta, and Yreka Creeks

Water temperature boundary conditions for the mainstem Shasta River at Dwinnell Dam, Parks Creek, the Little Shasta River, and Yreka Creek were estimated for unimpaired conditions according to equilibrium temperature theory (Martin and McCutcheon, 1999), using a spreadsheet model made in Microsoft Excel by Watercourse Engineering, Inc. Meteorological data from the California Department of Forestry's Brazie Ranch station was used to calculate hourly net heat flux at the air-water interface of a volume of specified dimensions. Net heat flux is the sum of solar radiation, atmospheric long wave radiation, long wave back radiation from the water surface, evaporative heat flux, and sensible heat flux. Hourly change in water temperature was then calculated using net heat flux, surface area, and given water properties such as density and specific heat capacity. Depth is user specified, and was set between 0.75 – 2 ft (0.23 – 0.61 m) to represent shallow tributary conditions. The governing equation for this model is a simplification of the advection diffusion equation:

$$\frac{\partial T_w}{\partial t} = S = \frac{q_n A_p}{C_p \rho V_p} \qquad \textbf{(48)}$$

where T_w is water temperature, t is hourly time step, S is sources/sinks, q_n is net heat flux, A_p is surface area, C_p is specific heat of water, ρ is density of water, and V_p is volume.

The equilibrium temperature model was calibrated to measured data at Parks Creek at the base of the mountains (NCRWQCB, 2004) (Figure 60, Figure 61). Measured data was used in place of the equilibrium temperature for periods where data were available (6/20 – 10/20). Original water temperatures were also adjusted to account for snowmelt influence from April 15 to July 15. A maximum decrease of 7°C was subtracted from water temperatures on June 1, with the snowmelt correction linearly moving to 0 from April 15th and to July 15th. (Similar approaches have been employed in the Trinity River basin, 2007.) The "final equilibrium temperature" in Figure 61 represents the boundary condition for Shasta River at Dwinnell Dam, Parks Creek, Little Shasta River, and Yreka Creek inflows.

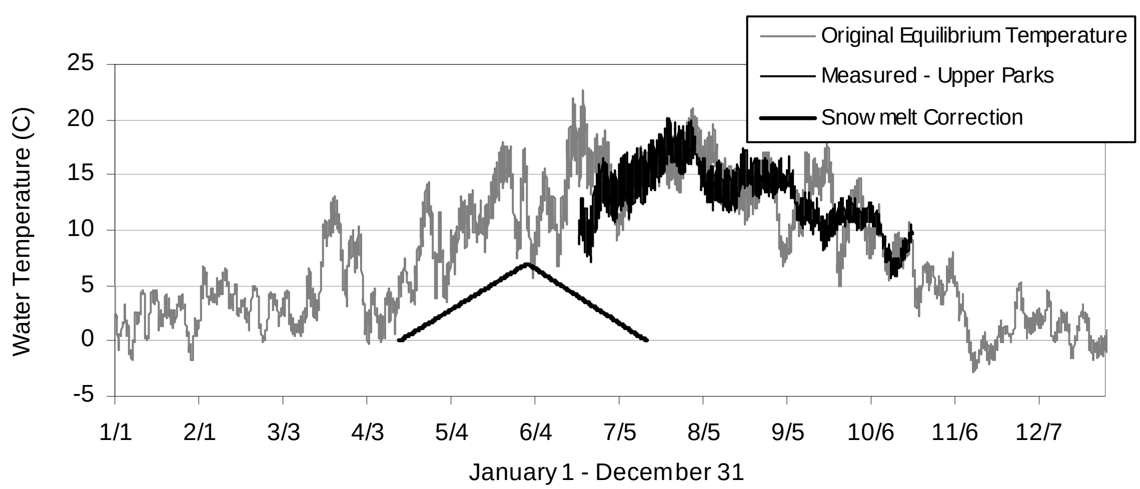


Figure 60. 2001 original equilibrium temperature trace with measured data and snowmelt adjustments

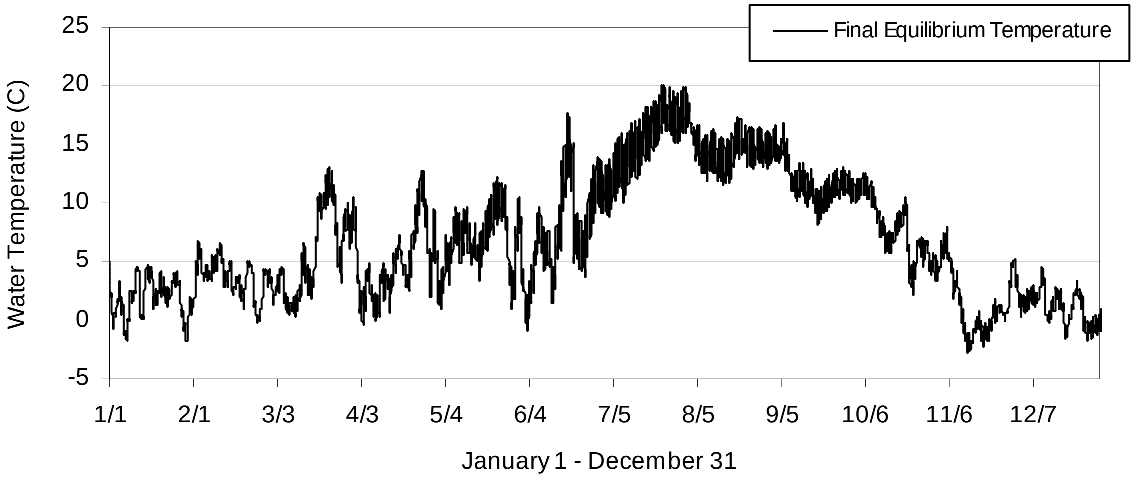


Figure 61. Final equilibrium temperature

Big Springs

The water temperature boundary condition for Big Springs was based on NCRWQCB data (NCRWQCB, 2004). A monthly average initial source temperature of 11.3°C was estimated at Big Springs, and the rate of heating (i.e., the change in temperature) over a 6-hour transit time to the Shasta River was estimated with the equilibrium temperature model (Figure 62). A 6-hour transit time was used as a conservative estimate. Travel time was calculated to be 2.2 hours using a water surface slope of 0.0015, Manning's n of 0.05, flow of 125 cfs, average reach velocity of 1.33 ft/s, and average width and depth of 75 ft (23 m) and 1.25 ft (0.4 m), respectively, with a rectangular channel assumption.

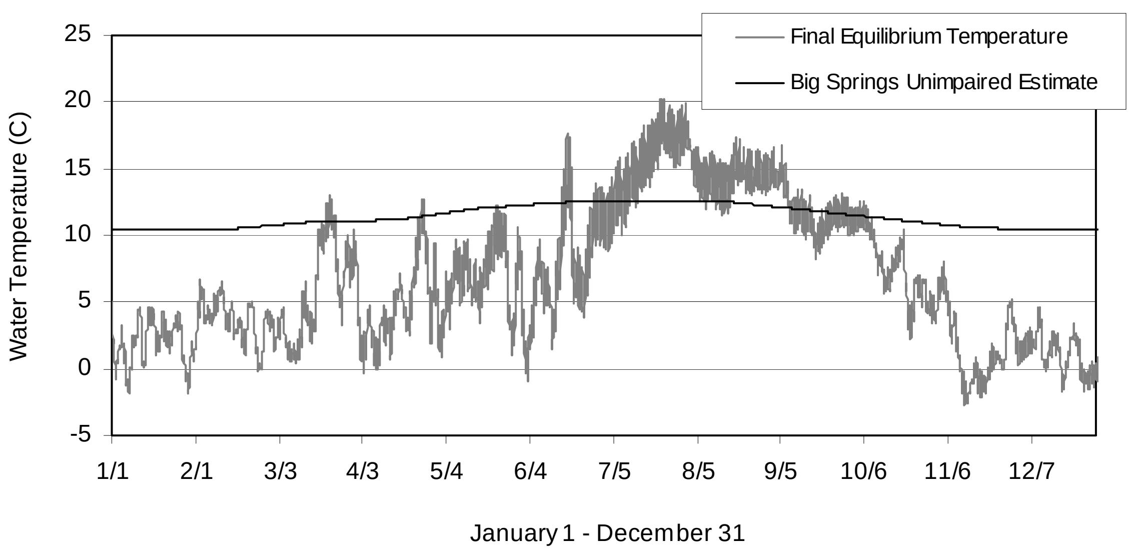


Figure 62. 2001 Big Springs unimpaired estimated water temperature

For the unimpaired model, it was assumed that Big Springs Creek is functional in terms of riparian vegetation, geomorphology in a dynamic equilibrium, groundwater connectivity, etc. The monthly distribution of Big Springs Creek water temperature at the Shasta River is listed in Table 13. Values for each month were assigned to the 15th of each month and linearly interpolated to create an hourly temperature boundary condition timeseries.

Table 13. Estimated unimpaired monthly distribution of water temperature for Big Springs Creek at the confluence with the Shasta River

Month	Temp (C)
January	10.40
February	10.50
March	11.02
April	11.16
May	12.02
June	12.47
July	12.55
August	12.54
September	11.89
October	11.22
November	10.57
December	10.42

Current Conditions

Hydrology

The current conditions model includes releases from Dwinnell Dam, the GID and SWUA diversions, and point source tributary inflows at Parks Creek, Big Springs, Little Shasta River, and Yreka Creek. Accretions and depletions representing numerous small diversions, tailwater return flow, and accretions from groundwater were modeled as distributed inflow at Big Springs to GID, GID to A12, A12 to Shasta River at Freeman (SRF), and the DWR weir to Anderson Road. Thus, current hydrology estimates were needed for all of the above-mentioned locations (Figure 63, Table 14). Input data were required for more reaches than unimpaired conditions because accretions and depletions (A/D) were applied to encompass the numerous small and moderate diversions along the Shasta River. 2001 was used to reconstruct a current conditions hydrology because it had the most measured data. The method for estimating flow is summarized in Table 15 and described below.

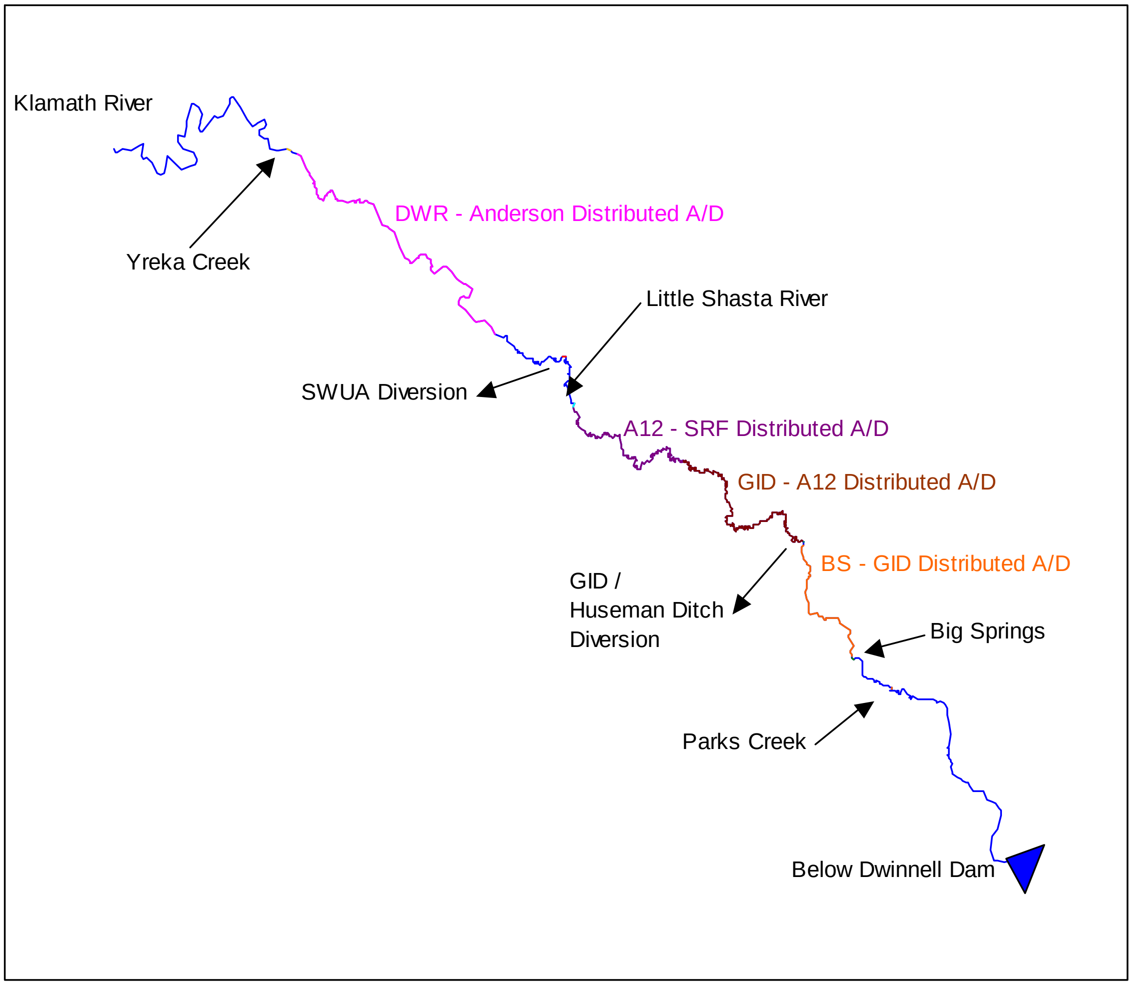


Figure 63. Current conditions distributed and point source inflows

Table 14. Major tributaries, diversions, and landmarks with river miles

Name	Type	River Mile
Dwinnell Dam	Dam (possible inflow)	40.62
Parks Creek	Inflow	34.92
Big Springs Creek	Inflow	33.67
GID/Huseman Ditch Diversion	Diversion	30.58
A-12 Road	Landmark	24.11
Little Shasta River	Inflow	19.19
SWUA Diversion	Diversion	17.85
DWR Weir	Landmark	15.52
Yreka - Anderson Road	Landmark	11.1
Anderson Grade	Landmark	8.1
Yreka Creek	Inflow	7.88

Table 15. Shasta River current conditions discharge estimation methods and data sources by reach

Reach	Dates	Data Calculation
Below Dwinnell	6/26 – 8/5, 8/15 – 9/21 1/1 – 6/25, 8/6 – 8/14, 9/22 – 12/31	Measured (NCRWQCB) 5.21% of SRM (CDEC)
Parks Creek	6/26 – 8/2, 8/15 – 11/16 1/1 – 6/25, 8/3 – 8/14, 11/17 – 12/31	Measured (NCRWQCB) – blw Dwinnell 5.21% of SRM (CDEC) + 1/3 of mouth – SRM if >0
Big Springs	6/1 – 9/30 1/1 – 5/24, 10/7 – 12/31 5/25 – 5/31, 10/1 – 10/6	Estimation of 70 cfs 60% of unimpaired estimate Linear ramping
Big Springs – GID	1/1 – 12/31	20% of SRM (CDEC)
GID / Huseman Diversion	5/1 – 9/1 4/1 – 4/30, 9/1/ - 9/30 1/1 – 3/31, 10/1 – 12/31	Estimation of -35 cfs Linear ramping 0
GID – A12	1/1 – 3/31 (75%) 4/1 – 10/14 (25%) 10/15 – 12/31 (75%)	Water balance (SRM-Blw Dwin-Parks-BS-GID-LS-SWUA) * seasonal percentage
A12 – SRF	1/1 – 3/31 (25%) 4/1 – 10/14 (75%) 10/15 – 12/31 (25%)	Water balance (SRM-Blw Dwin-Parks-BS-GID-LS-SWUA) * seasonal percentage
Little Shasta River	1/1 – 12/31	1% of unimpaired estimate + 1/3 of mouth – SRM if >0
SWUA Diversion	4/1 – 9/30 1/1 – 3/31, 10/1 – 12/31	Estimation of -42 cfs 0
DWR – Anderson	1/1 – 12/31	Water balance (Mouth –blw Dwin-Parks-Big Springs-GID-(GID-A12)-(A12-SRF)-LS-SWUA-Yreka)
Yreka Creek	1/1 – 12/31	1/3 of mouth – SRM if >0

Measured data for the reach below Dwinnell Dam was available from the North Coast Regional Water Quality Control Board (NCRWQCB). Measured hourly data was available from 6/26 – 8/5 and 8/15 – 9/21. During these time periods, flow at Parks Creek averaged 10.42% of the downstream flow at the Shasta River at Montague (SRM) gage

(CDEC, 2007). This percentage was split and 5.21% was applied as estimated releases or seepage below Dwinnell Dam and 5.21% was applied to Parks Creek for periods when no measured data exists (1/1 – 6/25, 8/6 – 8/14, 9/22 – 12/31).

Flow from Parks Creek was estimated using the same method as Dwinnell Dam. Measured data was available for Parks Creek from NCRWQCB from 6/26 – 8/2 and 8/15 – 11/16. 5.21% of the flow record at Montague (SRM) was used to estimate Parks Creek flow from SRM record when no recorded data exists (1/1 – 6/25, 8/3 – 8/14, 11/17 – 12/31). Additionally, one third of the difference between measured flow at the mouth and Montague was added when positive. (One third was also added to the Little Shasta River and Yreka Creek to capture storm peaks that normally make up a considerable portion of the flow in these creeks.)

Water rights on Big Spring Lake are 20 cfs during the irrigation season from 6/1 – 9/30 (Scott, pers. comm., 4/13/07). However, an estimated 40-50 cfs enters Big Spring Creek downstream of the lake (Deas et al., 2003). For this reason, Big Springs discharge was estimated to be 70 cfs from 6/1 – 9/30. From 1/1 - 5/25 and 10/7 – 12/31, Big Spring discharge was approximated to be 60% of the estimated Big Springs unimpaired flow (Deas, 2006). Discharge was ramped linearly between the 70 cfs summer estimate of Big Springs discharge and the winter season Big Springs flow calculation from 5/25 – 5/31 and 10/1 – 10/6.

In addition to all Big Springs and Parks Creek diversions, there are numerous diversions from the Shasta River above GID (CDWR, 2006). Distributed accretions and depletions from Big Springs – GID includes these diversions and return flows, as well as unquantified seepage, evaporation, subsurface, and overland flow along the upper Shasta River. 20% of measured flow at Montague (SRM gage) was applied to this reach to estimate accretions and depletions.

The GID / Huseman Ditch diversion was assumed to be 35 cfs from 4/1 – 9/30. GID has two pumps, when one pump is operating, approximately 20 cfs is diverted from the Shasta River; and when both pumps are operating approximately 40 cfs is diverted. Shasta Valley Watermasters estimate that two pumps operate approximately 85-90% of the time, thus 35 cfs is a reasonable estimate for a constant diversion (Scott, pers.comm., 2007). Linear ramping from 0 to 35 cfs was applied in the shoulders of irrigation season (4/1 – 4/30 and 9/1 – 9/30) because it fit measured data more accurately that beginning and ending diversions abruptly. From 1/1 – 3/31 and 10/1 – 12/31, GID / Huseman Ditch diversions are zero.

Accretions and depletions for the GID to A12 and A12 to Shasta River at Freeman reaches were estimated by water balance (A/D at these reaches is the excess/required water to make the water balance close from Dwinnell Dame to the DWR weir. From 1/1 – 3/31 and 10/15-12/31, 75% of A/D is distributed at GID-A12, and 25% at A12-SRF. From 4/1 – 10/14, 25% of A/D is distributed at GID-A12, and 75% at A12-SRF. This implies that during irrigation season, the GID-A12 reach may have relatively more diversion, and the A12-SRF reach may receive more return flow. Outside of irrigation season, diversions and return flow decrease, and more natural seepage/runoff/springflow may occur at the GID-A12 reach.

Flow for the Little Shasta River was calculated similarly to that of Big Springs. Shasta Valley Watermasters estimate discharge from the Little Shasta River is approximately 1 cfs from 6/1 – 9/30 and flow remains low throughout the year (Scott, pers.comm., 2007). Thus, Little Shasta flow was estimated to be 1% of the estimated unimpaired flow at the Little Shasta River, plus one third of the difference between measured flow at the mouth and Montague when the difference was positive.

Shasta Valley Watermasters estimate that the Shasta Water User's Association (SWUA) diverts 42 cfs from 4/1 – 9/30. During the rest of the year, SWUA diverts no water from the Shasta River.

All flows were summed so that accretions and depletions in the DWR weir to Anderson Rd reach closed the water balance. The water balance was calculated as flow at the mouth minus downstream Yreka Creek and all upstream reaches (below Dwinnell Dam, Parks Creek, Big Springs, GID, GID – A12, A12 – SR at Freeman, the Little Shasta River, and the SWUA Diversion). On average, distributed flow to this reach was approximately 50% of the measured flow from the Shasta River at Montague (SRM) gage.

Yreka Creek was estimated as one third of the difference between measured flow at the mouth and Montague when the difference was positive. From Yreka Creek to the mouth, no inflow, outflow, accretions, or depletions were assumed. Recorded flow data from the Shasta River at Yreka (SRY) was used to confirm the water balance for all the upstream reaches of the Shasta River (CDEC, 2007). The sum of all inflows and outflows from all reaches equaled measured data from the Shasta River at Yreka (SRY) gage. For all locations, hourly records were aggregated to average daily flow (Figure 64). The downstream boundary condition remained unchanged from the unimpaired model, in which a daily elevation hydrograph set downstream elevation to 2,042 ft (622 m) above sea level.

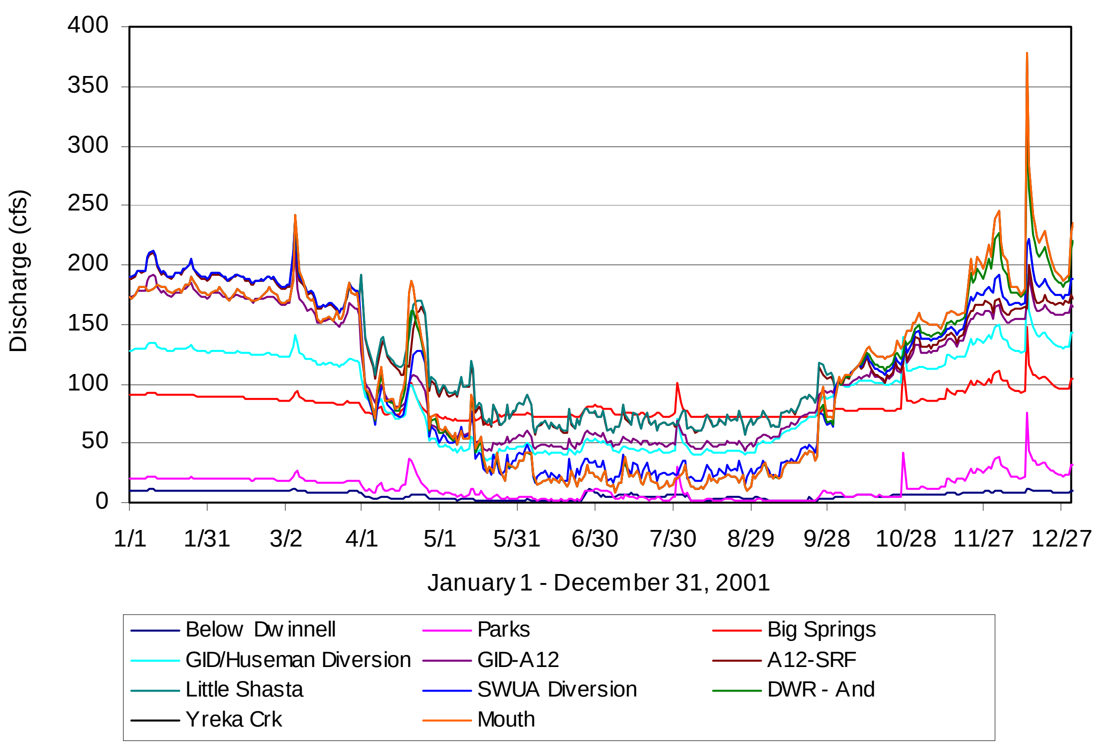


Figure 64. Current conditions daily flow timeseries

Direct measurement of tributaries would greatly improve model results, especially at Big Springs, which contributes relatively high, constant flows to the system. Without direct measurement at tributaries, flows were estimated from downstream records. Also, small, individual diversions and all tailwater return were lumped into accretions and depletions. Improved diversion and return flow estimates and schedules could improve the model.

Water Temperature Boundary Conditions

Water temperature boundary conditions were needed for Dwinnell Dam, Parks Creek, Big Springs, Little Shasta River, and Yreka Creek. Other reaches with diversions, accretions, or depletions used simulated temperature at those locations rather than user specified temperatures. Like the unimpaired model, an equilibrium temperature model was used to estimate water temperature for periods or locations where temperature records were not kept. The temperature equilibrium model was calibrated to measured data, when available.

Measured water temperature above Parks Creek was available from NCRWQCB from 4/24 – 10/13/2001, and was used to calibrate equilibrium water temperature below Dwinnell Dam (Figure 65). The measured data were used as model input when available. Average water depth was assumed to be 0.75 ft (0.23 m) in the equilibrium temperature model. A maximum of 7°C was added to the equilibrium temperature pattern on 1/1/01, and was linearly ramped to 3°C by 5/1/01. Similarly 3°C was added to all equilibrium temperature signals on 10/1/01, and was linearly ramped to 7°C by 12/31/01. This was done for all boundary condition temperature signatures because boundary condition temperature signatures hovered near 0°C during winter (Jan, Feb, Mar, and Dec). Water temperature between 0 - 5°C are probably too cold for the Shasta River as it is spring-derived (meteorological conditions drive water temperature in the equilibrium temperature model). Thus temperature output from the temperature equilibrium model was increased during winter.

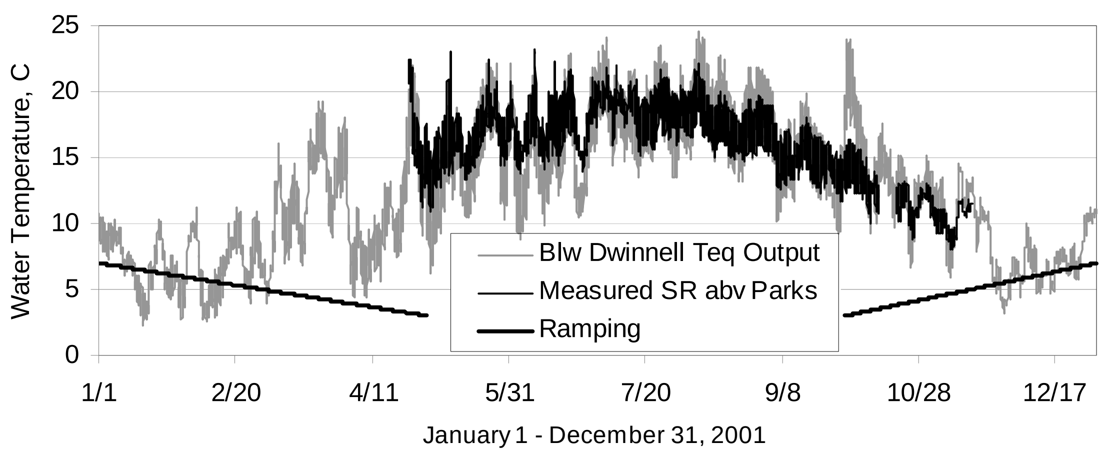


Figure 65. Below Dwinnell Dam equilibrium temperature trace with measured data

Measured data for Parks Creek in 2001 was available from NCRWQCB from 4/24 – 10/13/2001. These data were used directly as the Parks boundary condition, and the equilibrium temperature model was used to estimate Parks Creek temperature for periods when water temperature was not recorded at this site (Figure 66). In the temperature equilibrium model, water depth was assumed to be 0.3 ft (9 cm), and minimum water temperature was set to 5°C. The final Parks Creek temperature trace used linear ramping to increase water temperature during winter months.

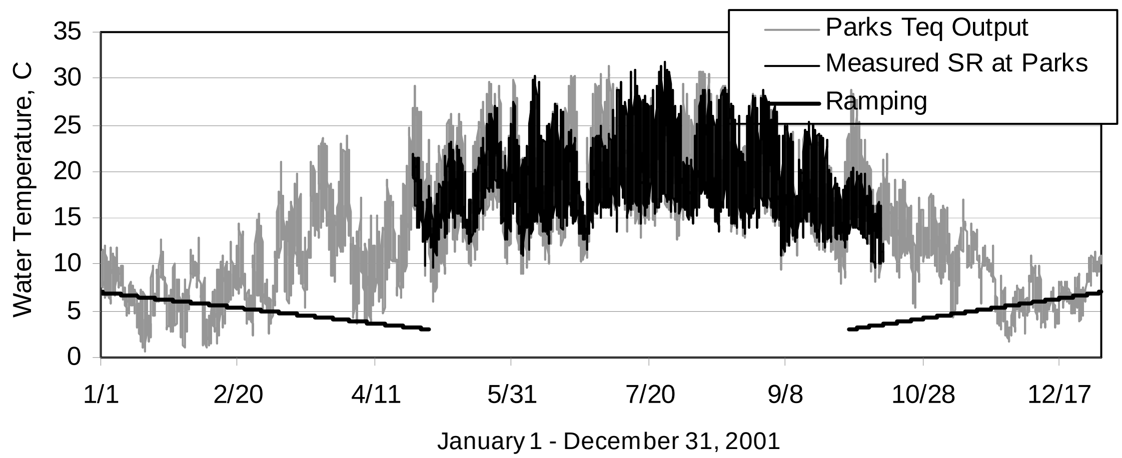


Figure 66. Parks Creek equilibrium temperature trace with measured data

Measured water temperature data from the Shasta River at GID (NCRWQCB, 2004) was applied to Big Springs. No temperature data exists for Big Springs and inflows above Big Springs are small compared to inflow from Big Springs; thus, the temperature at GID should resemble that of Big Springs. Measured data was used for the temperature boundary condition, for summer months (5/24 – 11/16), and the equilibrium temperature model was used to estimate temperature data when it was not measured directly (Figure 67). Minimum temperature was also constrained to 2°C, and water depth was assumed to be 1.0 ft (0.3 m). Big Springs originates approximately two miles from the Shasta River. At its source, Big Springs has a constant temperature of approximately 11.3°C (NCRWQCB, 2004). During summer months, water from Big Springs heats considerably before reaching the Shasta River. To preserve relatively warm winter temperatures, temperature was increased by 3-7°C, as it was with Dwinnell releases and Parks Creek. Finally, a 7-day average of the equilibrium temperature at Big Springs was used for the final Big Springs boundary condition. This increased daily thermal variability and decreased weekly and monthly temperature extremes.

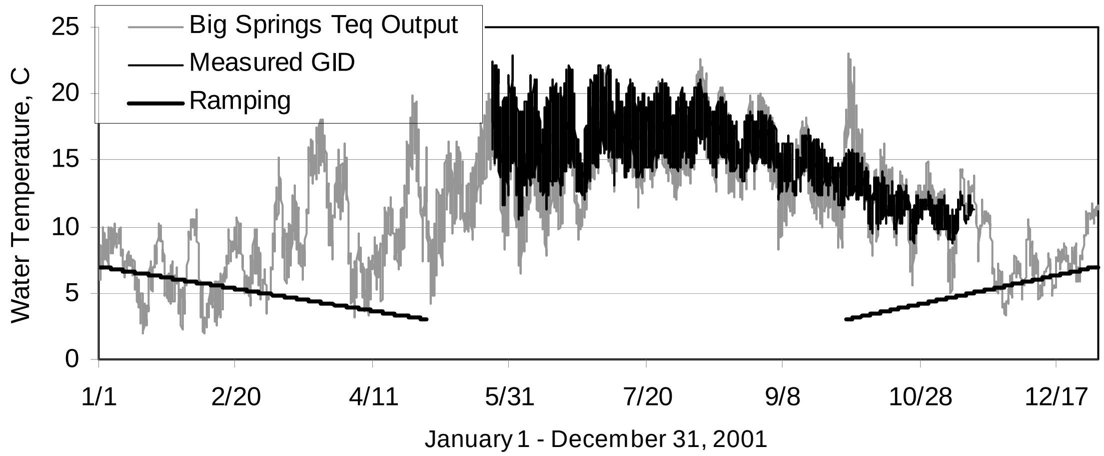


Figure 67. Big Springs equilibrium temperature trace with measured data

Measured data for the Little Shasta River was only available in 2003 (NCRWQCB, 2004); thus the equilibrium temperature model was used to estimate the Little Shasta temperature boundary condition, but was not used in the boundary condition temperature data (Figure 68). Since discharge from the Little Shasta River is generally low, water depth was assumed to be 0.4 ft (12 cm) and initial water temperature was 5°. Minimum water temperature was constrained to 2°C, as with other Shasta River tributaries. The Little Shasta River temperature trace was also used as the Yreka Creek boundary condition.

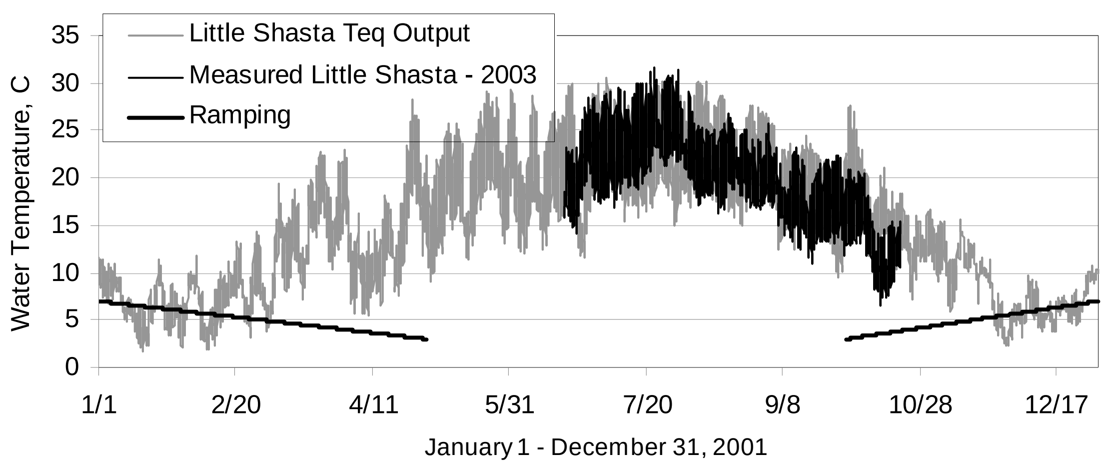


Figure 68. Little Shasta River equilibrium temperature trace with measured data

Meteorology

Meteorological input includes: cloud cover (Figure 69), dry bulb temperature (Figure 70), dew point temperature (Figure 71), wind speed (Figure 72), short wave solar radiation (Figure 73), and elevation based atmospheric pressure which was constant at

930.41 mb. Meteorological data is identical for all simulations. Dry bulb temperature, atmospheric pressure, wind speed, and solar radiation were obtained from California Department of Forestry's Brazie Ranch station (CDEC). Dew point temperature was calculated as:

$$T_{dp} = 237.3\left(\frac{B}{1-B}\right) \quad \textbf{(49)}$$

$$B = \ln\left(\frac{e}{6.108}\right)/17.27 \quad \textbf{(50)}$$

$$e = RH * \frac{e_s}{100} \quad \textbf{(51)}$$

$$e_s = 6.108\exp\left[\frac{17.27T_a}{(T_a + 237.3)}\right] \quad \textbf{(52)}$$

where T_{dp} is dew point temperature (C), RH is relative humidity (%), e is vapor pressure (mb), e_s is saturation vapor pressure (mb), T_a is dry bulb temperature (C) (Chapra, 1997).

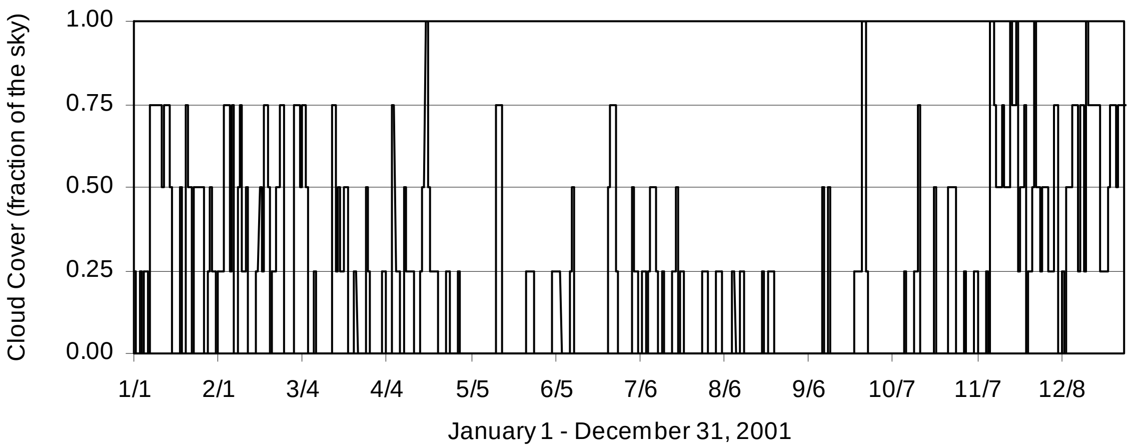


Figure 69. Cloud cover as a fraction of the sky (0-1)

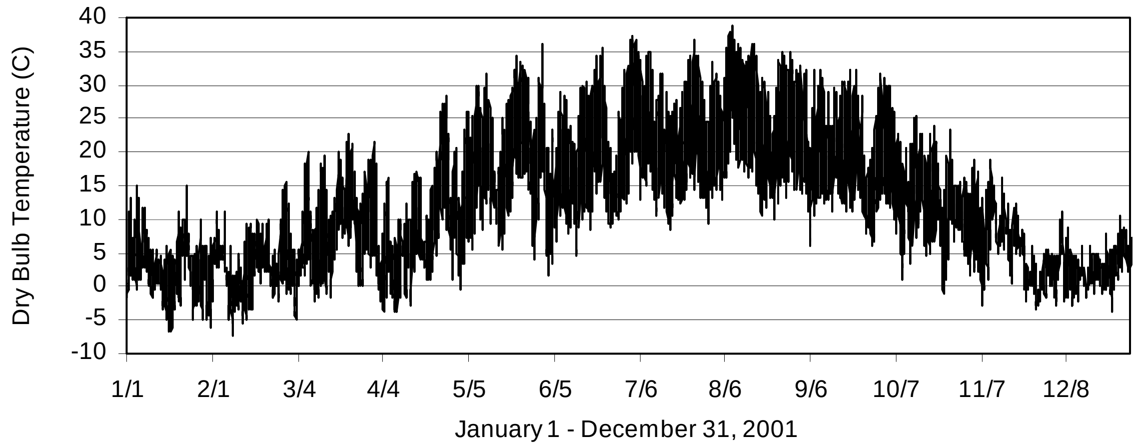


Figure 70. Dry bulb temperature (C)

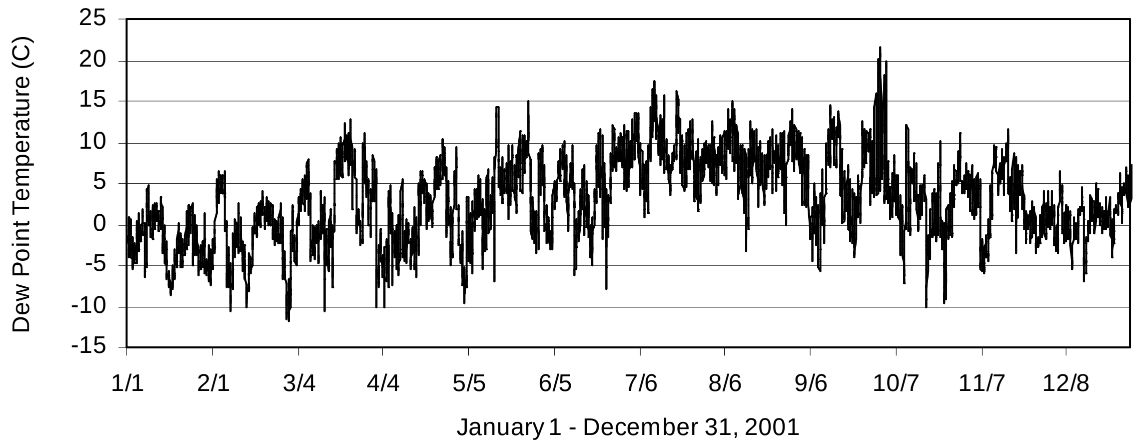


Figure 71. Dew point temperature (C)

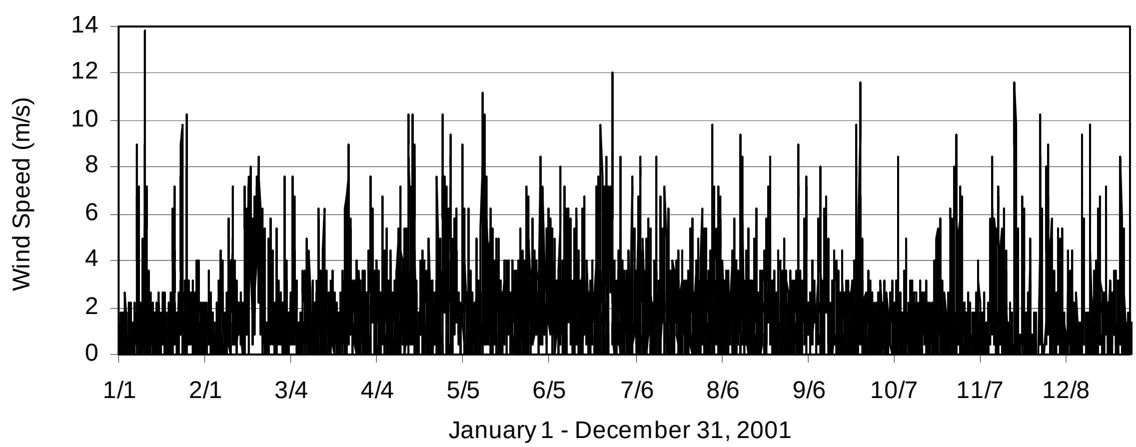


Figure 72. Wind speed (m/s)

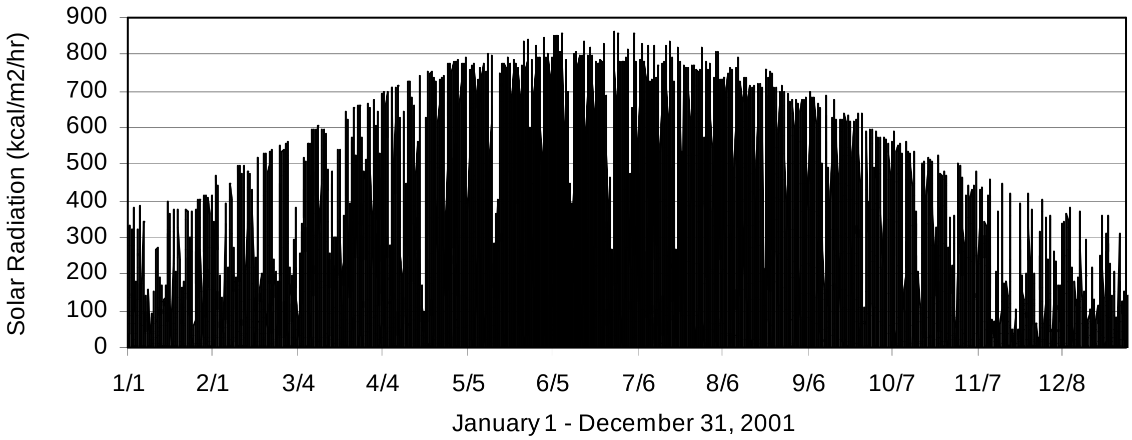


Figure 73. Short wave solar radiation (kcal/m^2/hr)

Riparian Shading

Riparian shading data was taken directly from the TMDL model. Tree height was 22 ft (6.7 m) with variable transmittance densities as specified in Figure 74 (1 = no shade, 0 = full shade). Lowney (2000) estimates deep riparian foliage may have a transmittance of 10-20%, with remaining solar radiation absorbed or reflected by vegetation. However, riparian vegetation sampling completed in 2002 indicates vegetation along the Shasta River is not continuous and does not form a complete canopy. For this reason, 50% is used as a maximum bound of solar transmittance as a conservative estimate. Riparian shading is the same for all simulations, except the riparian shading alternatives.

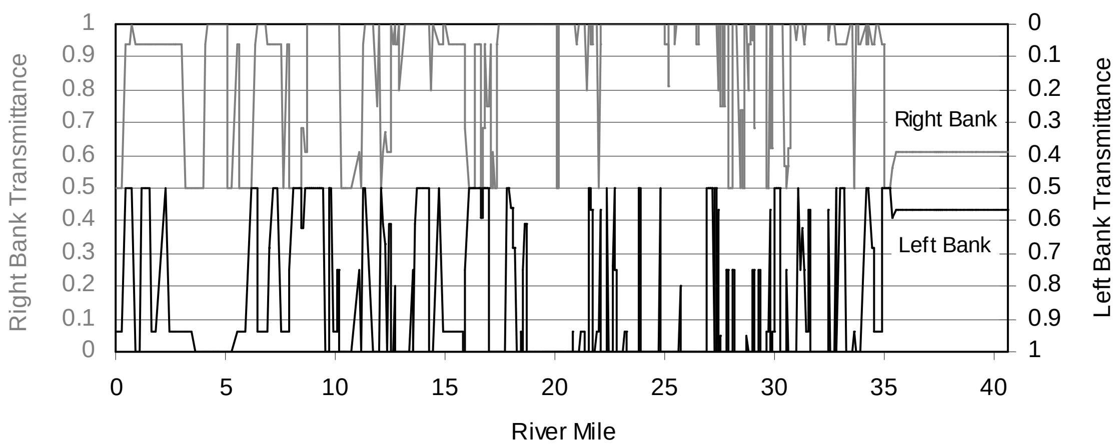


Figure 74. Riparian Shading transmittance model input for left and right banks

Model Testing

The year long model was re-calibrated because previous calibrations were only for a few weeks during summer (Abbott, 2002; Deas et al., 2003; Geisler, 2005). This section describes calibration steps, including modifications to the geometry file, and changes to the Manning's n and water quality parameters. This section ends with comparisons of simulated and measured flow and water temperature at all locations with measured data in 2001, and accuracy of fit statistics such as mean bias, mean absolute error, and root mean square error. Mean absolute error was less than 10 cfs and 2°C for all sites with measured data.

The geometry above Anderson Road (RM 8.05 – 12.00) was narrowed to 40% of the original. Cross-section distance was measured at five locations in this reach in 2001 (Abbott, 2002). One location was at Anderson Rd Bridge, which is not representative of the channel in this reach. The channel narrows immediately upstream and downstream of the bridge. Two additional cross-sections were measured on the Peters' property and two on the Fiock's property. The dam at Yreka-Ager Road at RM 10.91 was removed and riparian vegetation may have encroached on the channel, both of which could alter channel morphology and width. The system was sensitive to geometry, suggesting a reduction was appropriate. Original channel width in this reach was approximately 55-60 ft (16.8 – 18.3 m), and was reduced to 22-24 ft (6.7 – 7.3 m).

River geometry was also narrowed by 60% above Louie Rd (RM 33.93 – 35.82). One cross-section was measured above Parks Creek (RM 35.4) in 2001 (Abbott, 2002). In general, this is a poorly defined reach with very low flow (approximately 10 cfs). Original channel width was 29 ft (8.8 m) at river mile 35.82, widening to 40 ft (12.2 m) at river mile 33.93. Channel width was narrowed to 17 ft (5.2 m) to 24 ft (7.3 m), respectively. Reducing channel width above Anderson Rd and Parks Creek reduced daily temperature variability to better represent measured data.

At the GID diversion, Manning's n was changed from 0.05 to 0.3 to simulate the diversion dam and upstream ponding. Additionally, two water quality parameters were changed. The thermal diffusivity of bed material was changed from the recommended value of 27.7 cm^2/hr to 25 cm^2/hr, which remains in the recommended range (Hauser and Schohl, 2002). This reduced thermal variability in the model. Also the wind coefficient in wind-driven evaporative cooling was changed from 1.0E-09 m^3/mb/s, the value in previous Shasta River models, to 0.5E-09 m^3/mb/s, the recommended value (Hauser and Schohl, 2002). This raised water temperature by approximately 0.5 - 1°C. Additional adjustments were tested, such as changing the values for the wind-exponent in wind-driven evaporative cooling, light extinction coefficient, upper layer bed thickness, and maximum multiplier on Manning's n at shallow depths, but ultimately were left unchanged.

Simulated flow and water temperature were compared to measured data where available to test model accuracy (Table 16, Figure 75 - Figure 80). Mean absolute error was less than 10 cfs for all sites with measured data. Discharge at Parks Creek, the DWR weir, and the mouth of the Shasta River matched measured data well, although storm runoff periods were sometimes not captured in simulations, probably from aggregating hourly data to daily data. Modeled flow was approximately 15 cfs too low during summer at GID, and was approximately 25 cfs too low for one month after irrigation season ends at Anderson Road.

Table 16. Measured versus modeled flow statistics

	Mean Bias cfs	MAE cfs	Average measured flow cfs	RMSE cfs	n
Parks	0.98	1.16	5.60	3.59	3129
GID	-4.60	7.57	79.37	9.59	2854
A12	-1.99	7.39	81.32	9.52	3044
DWR Weir	0.00	2.42	104.95	4.03	8738
Anderson	-4.58	8.42	65.49	11.95	3241
Mouth	0.06	3.17	106.89	7.10	8737

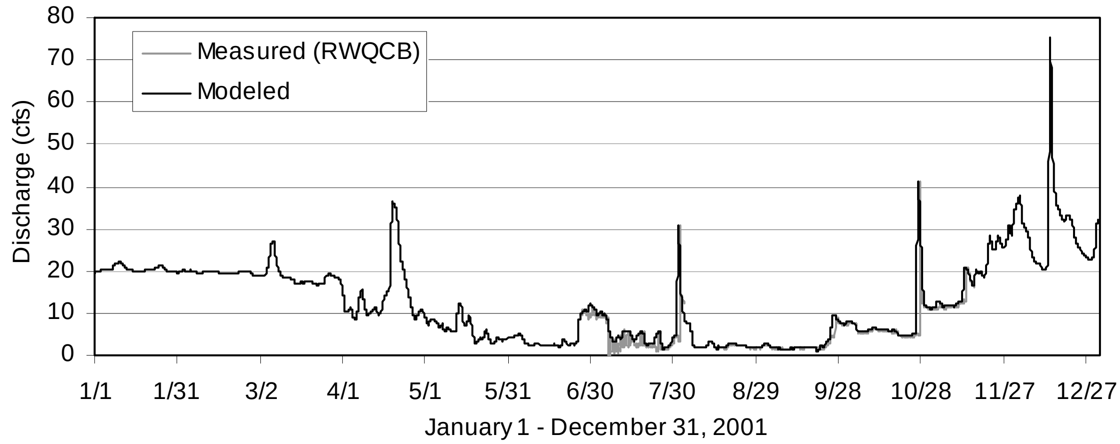


Figure 75. Parks Creek measured versus modeled flow

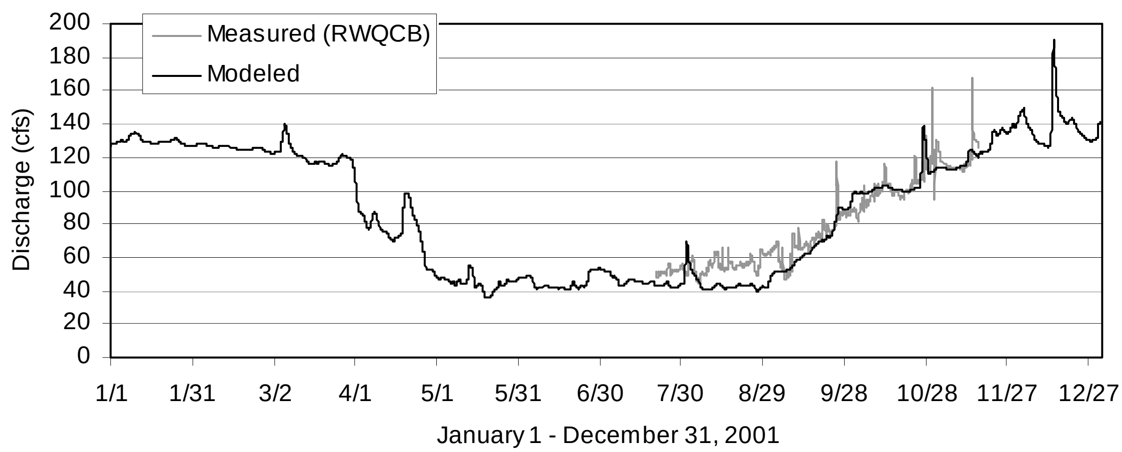


Figure 76. GID measured versus modeled flow

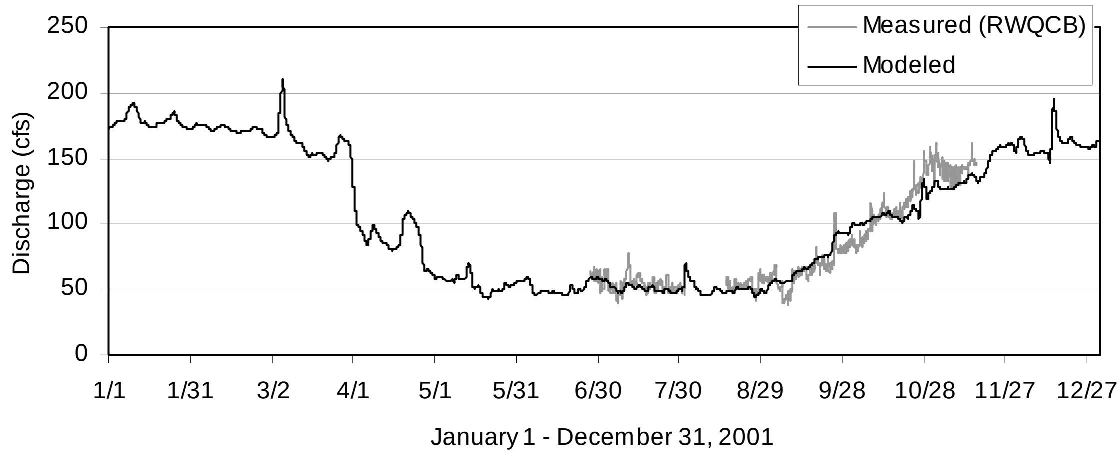


Figure 77. A-12 measured versus modeled flow

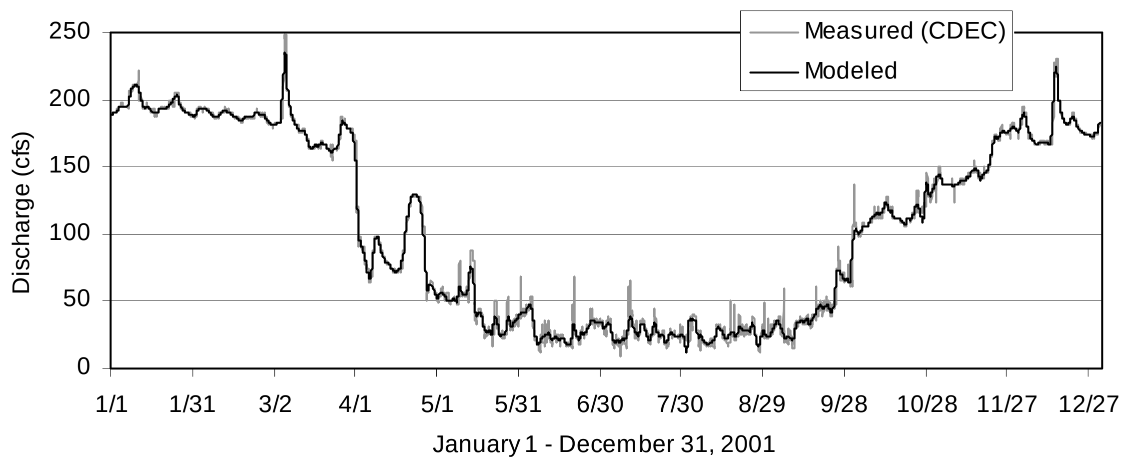


Figure 78. DWR weir measured versus modeled flow

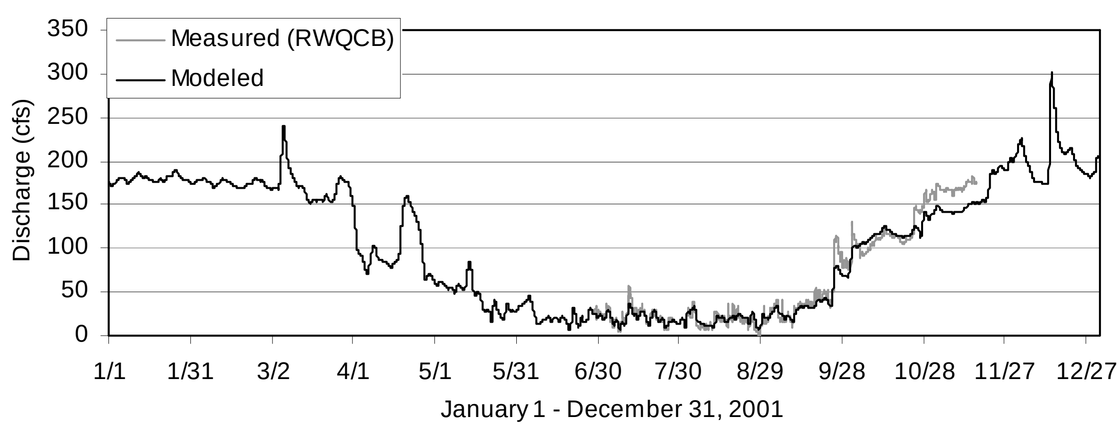


Figure 79. Anderson Road measured versus modeled flow

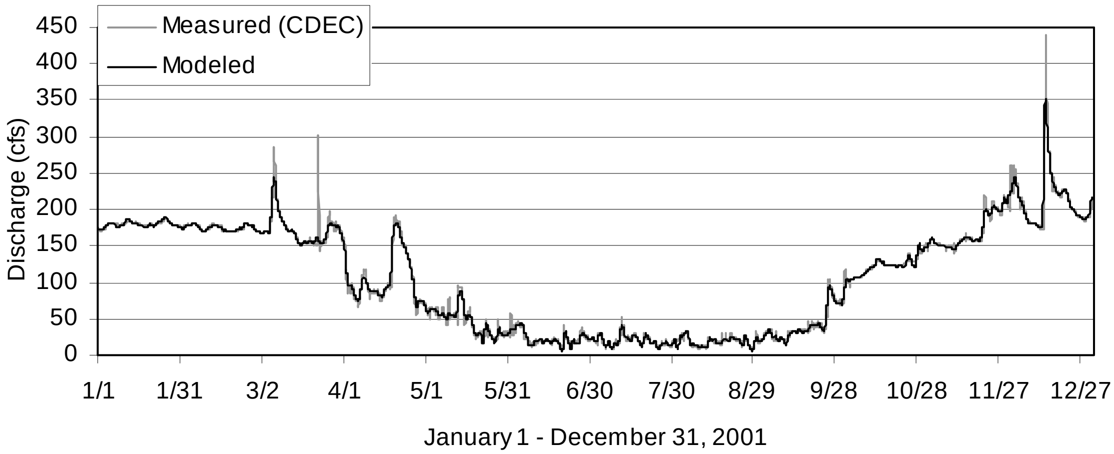


Figure 80. Mouth measured versus modeled flow

Mean absolute error of modeled data was within 2°C at all locations with measured water temperature (Table 17, Figure 81 - Figure 89). Measured data above Parks Creek was used directly for the water temperature boundary condition below Dwinnell Dam. The modeled diurnal signal at Parks Creek is too narrow, at Louie Rd is too wide, and at GID is again too narrow (and at the upper end of the measured data). Since modeling results did not systematically have the same problem (i.e., consistently too warm or too cool), more accurate calibration was difficult without additional input data or increased knowledge of the system. Timing of daily temperature variations match measured data well, except at GID, where the modeled signal can be 2-4 hours earlier than the measured signal.

Simulated water temperatures were colder during the winter than measured temperature from the Shasta River at all sites. RMS numerical models can underpredict temperatures during winter when water temperature is below 5-10°C (Deas, pers.comm., 2008). Furthermore, RMS is driven by meteorological conditions, although the Shasta River is influenced by both springflow and meteorological conditions, particularly near Big Springs. Water temperature is not critical during winter, so this is not a significant deviation from measured water temperature for the purposes for which this model is being applied.

Table 17. Measured versus modeled water temperature statistics

	Mean Bias C	MAE C	RMSE C	n
Abv Parks	0.00	0.00	0.00	4798
Parks Crk	-0.96	1.48	2.00	4125
Louie Rd.	-0.09	1.90	2.27	6471
GID	0.57	1.82	1.31	4224
A12	-0.44	1.29	1.66	7668
DWR Weir	-0.47	1.30	1.62	7670
Hwy 3	-0.15	1.40	1.72	4177
Anderson	-0.70	1.34	1.65	7671
Mouth	-0.98	1.73	2.07	8461
Shasta Average	***-0.36***	***1.36***	***1.59***	***6141***

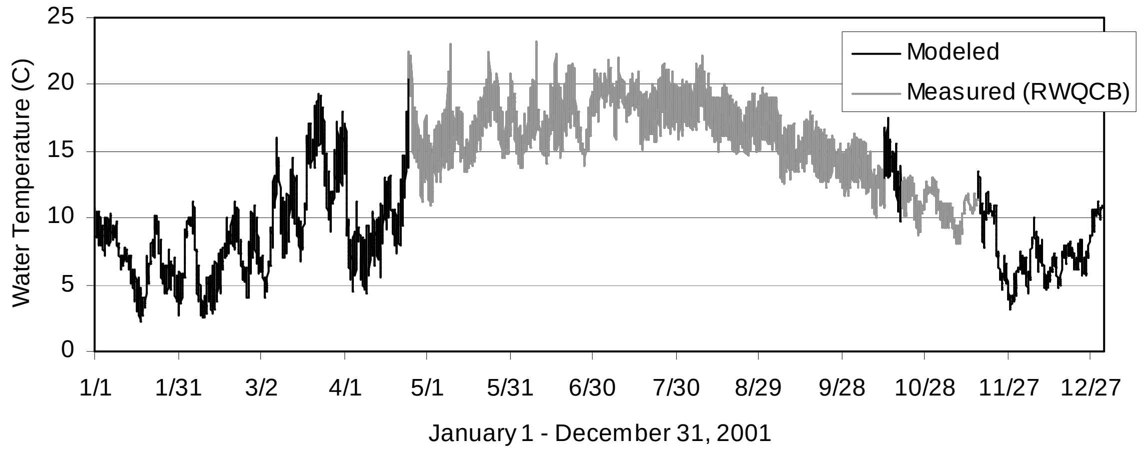


Figure 81. Above Parks Creek modeled versus measured water temperature

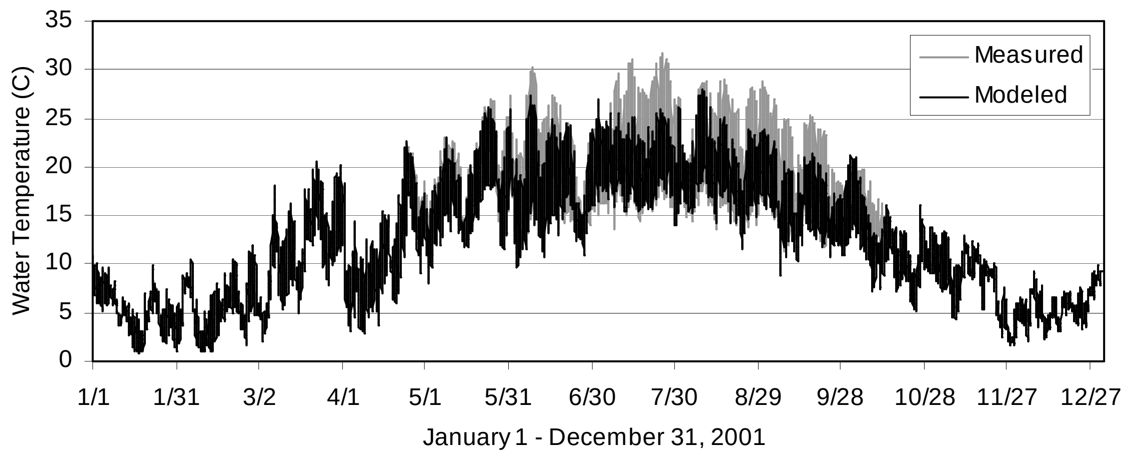


Figure 82. Shasta River at Parks Creek measured versus modeled water temperature

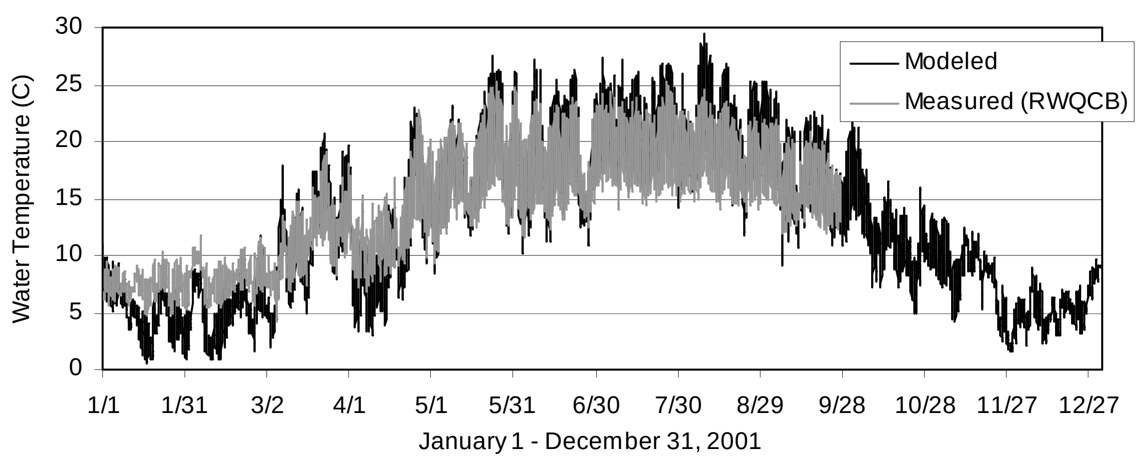


Figure 83. Shasta River at Louie Road modeled versus measured water temperature

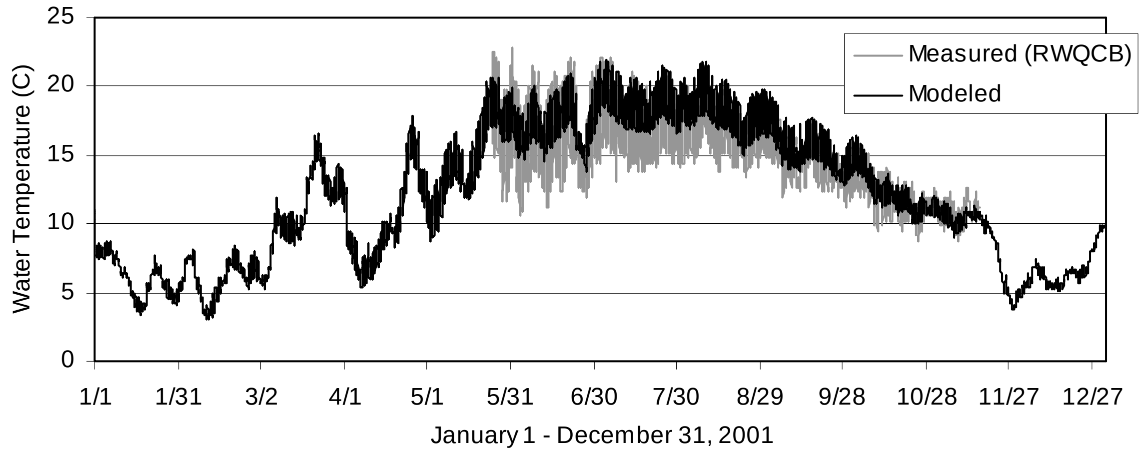


Figure 84. GID modeled versus measured water temperature

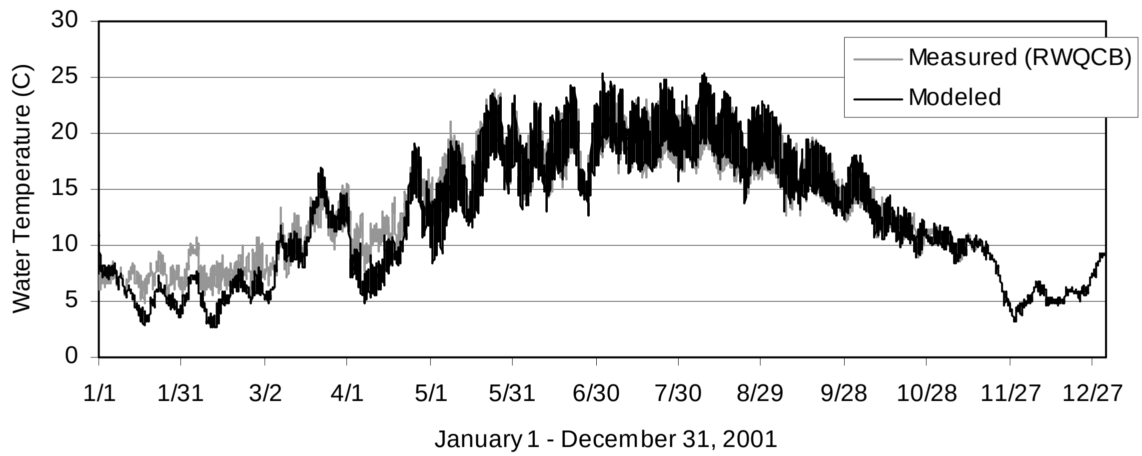


Figure 85. A12 modeled versus measured water temperature

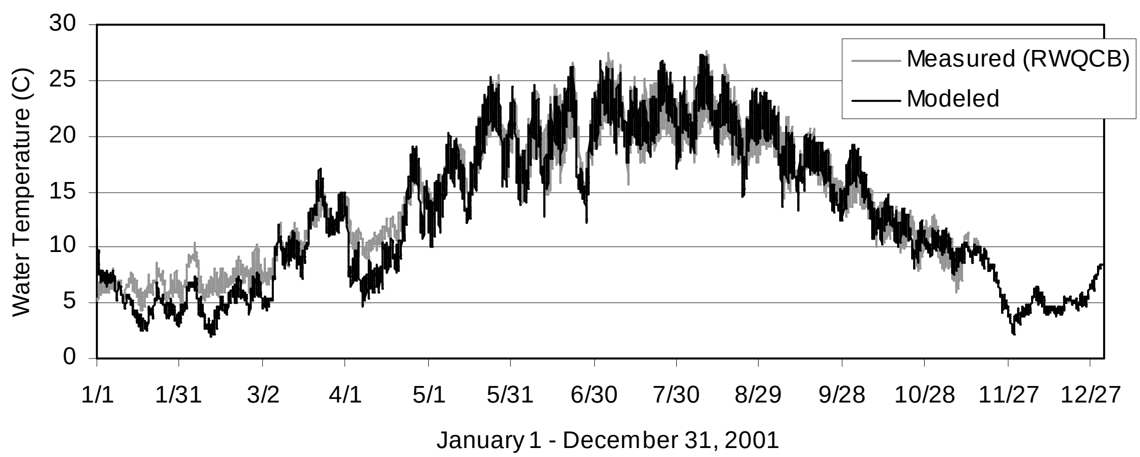


Figure 86. DWR weir modeled versus measured water temperature

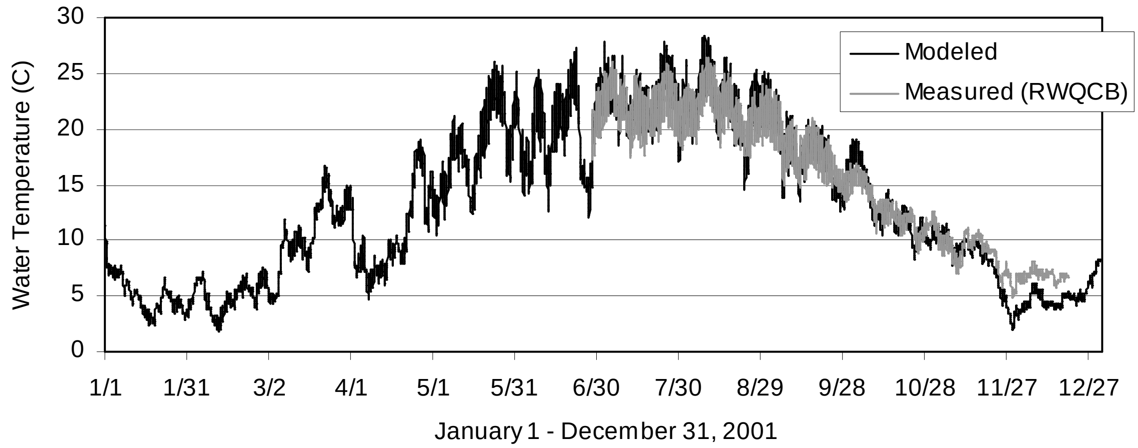


Figure 87. Hwy 3 modeled versus measured water temperature

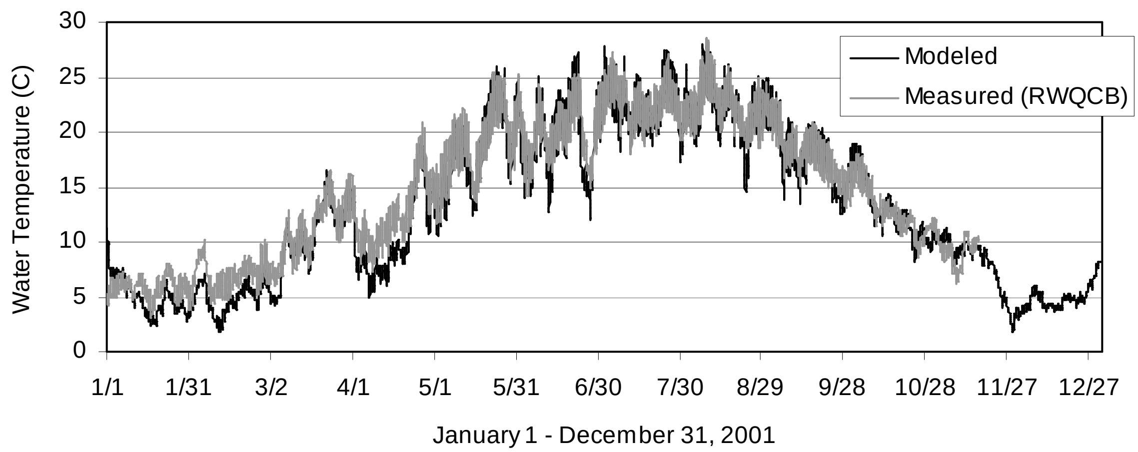


Figure 88. Anderson Road modeled versus measured water temperature

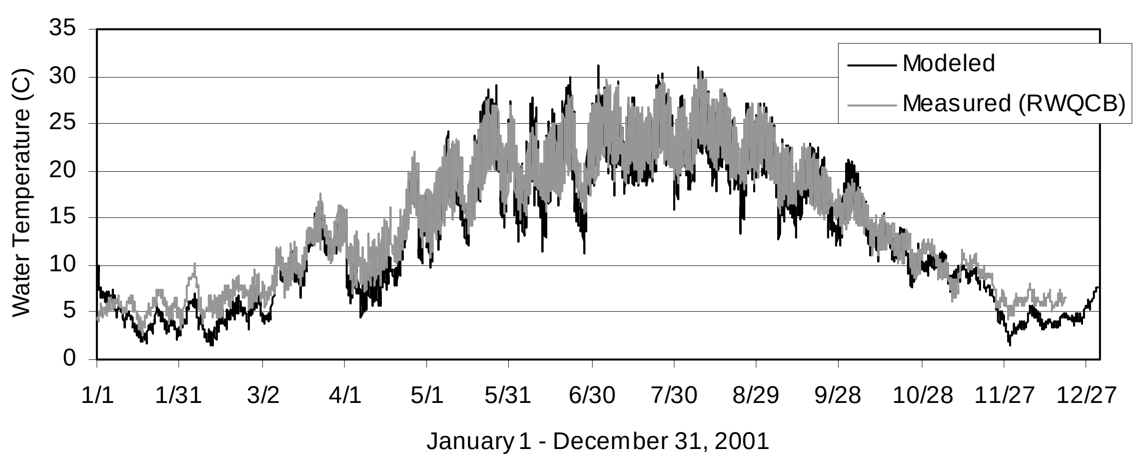


Figure 89. Mouth modeled versus measured water temperature

Results

Model runs were completed with the two modeling sets described above to analyze eight restoration alternatives:

- Unimpaired conditions
- Current conditions
- Shasta River minimum instream flows
- GID diversion alternatives
- Nelson Ranch return flow analysis
- Riparian vegetation alternatives
- Restore Big Springs complex
- Remove Dwinnell Dam

Unimpaired and current conditions provide bookends for the potential range of flows and water temperatures in the Shasta River. Some alternatives, such as return flow analysis and riparian vegetation modeling explore how restoration decisions may affect water temperature to increase understanding and guide local management decisions.

Unimpaired Conditions

The unimpaired run simulates conditions prior to water and land development in the Shasta River basin. It represents conditions without Dwinnell Dam, groundwater pumping, water diversions, or tailwater return flow, and includes moderate riparian shading and a small diurnal signal at Big Springs representing channelized, shaded flow prior to the confluence with the Shasta River.

Flow input data was averaged monthly, so model output has no storm-related pulses (Figure 90). Historic data suggests winter baseflow exceeded 300 cfs (Deas et al., 2004; CDWR Watermaster service records, 1930-1990), and pulses greater than 500 cfs probably occurred following storms. The larger, more consistent flow would increase the incidence of floodplain inundation during high flow events in winter and spring, opening floodplain and side channel habitat for young salmon emerging from redds and rearing in the Shasta River. The large stable inflow from Big Springs kept baseflow above 150 cfs downstream of the Big Springs complex throughout summer. Yearly low flow conditions on the Shasta River would have occurred in early autumn, like most rivers in California.

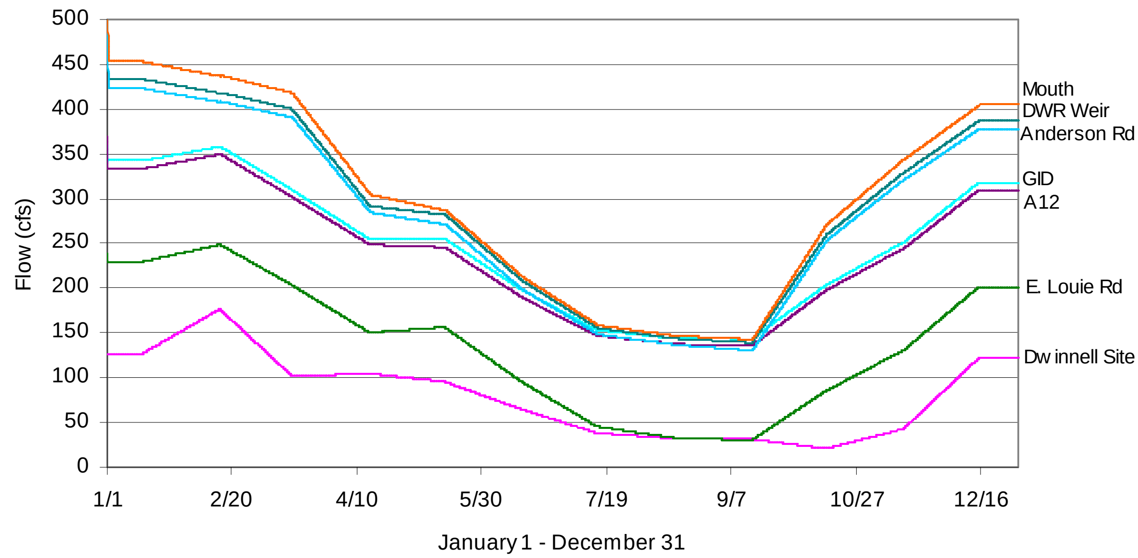


Figure 90. Modeled unimpaired flow for select Shasta River locations

Simulated winter water temperature is probably a low estimate, and may have been between 5-10°C from the temperature moderating effect of springflow (Figure 91). During spring and fall, Big Springs may have had only a modest effect on water temperature because equilibrium temperature was close to the temperature of the springs from mid-September to late October, and April to May. During summer, solar radiation heated the Shasta River from Big Springs to the mouth, although riparian vegetation and increased thermal mass reduced this effect. Summer water temperature may have remained well below 20°C at GID, and below 25°C at the mouth. At GID, minimum water temperature remained below approximately 13°C, providing relief for fish following warm, summer days (Figure 92).

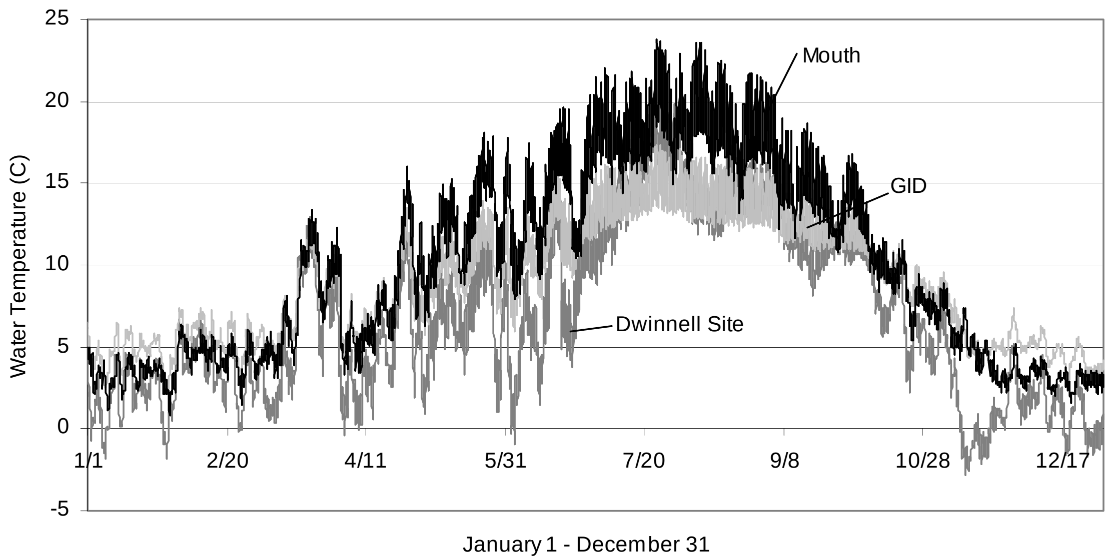


Figure 91. Modeled unimpaired water temperature for select Shasta River locations

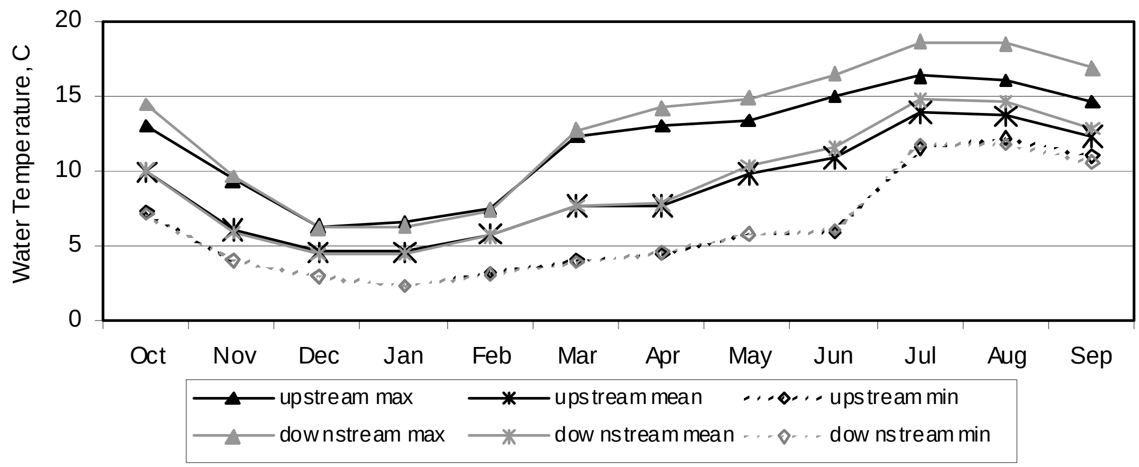


Figure 92. Simulated unimpaired max, mean, and min water temperature at Nelson Ranch upstream and downstream property boundaries

Current Conditions

During winter, baseflow exceeds 100 cfs at GID, and 150 cfs in the lower Shasta River (Figure 93). In all but the wettest years, diverted Parks Creek flows and water from the Shasta River above Dwinnell Dam are stored in Lake Shastina. The Shasta River has only small peaks from local storm events and runoff from the Big Springs complex. Dwinnell spills infrequently during wet years (i.e. 1964 and 1997) (Jeffres et al., 2008), and during these years Parks Creek is not diverted to the reservoir. Large storm pulses no longer occur because of water development in the Shasta basin. Summer has extreme low flow conditions in the lower river, with flow consistently below 50 cfs from mid-May to late-September. No water is released from Dwinnell Dam, except to fulfill downstream water rights. Flows from springs to the Shasta River are remarkably resilient, with baseflow increasing as soon as irrigation season ends. During the first week of October, stages increase markedly.

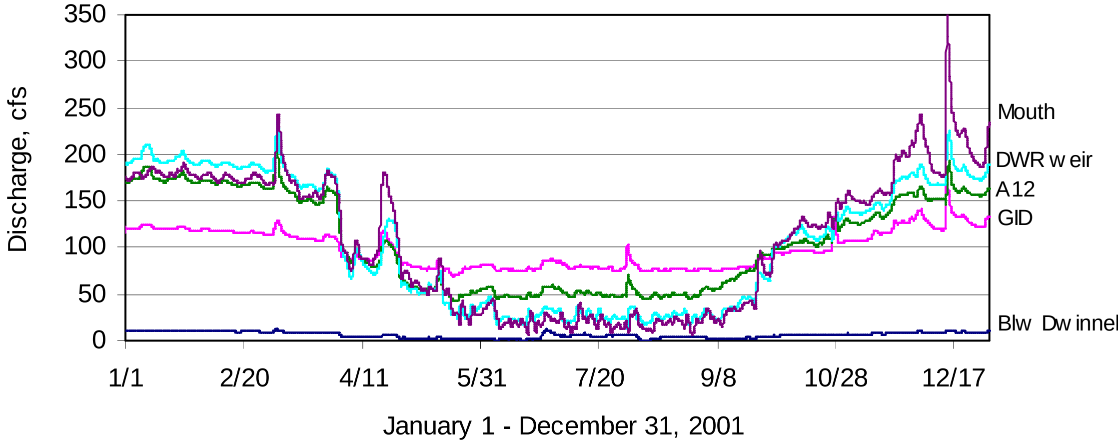


Figure 93. Modeled current conditions flow for select Shasta River locations

Like the unimpaired model, winter water temperatures are between 5-10°C from springfed contributions (Figure 94). Water temperature of the Shasta River exceeds springflow temperature by mid-April. Although Big Springs remains a cool-water input to the Shasta River through spring and summer, it no longer is channelized (as it was under unimpaired conditions) and is exposed to solar radiation, increasing the temperature and diurnal range of the springflow. Big Springs stabilizes temperatures; although, less riparian shading and less thermal mass allow rapid heating over the length of the Shasta River (Figure 95). Water temperature at GID is now well above 20°C, and temperatures at the mouth reach 30°C (Figure 94).

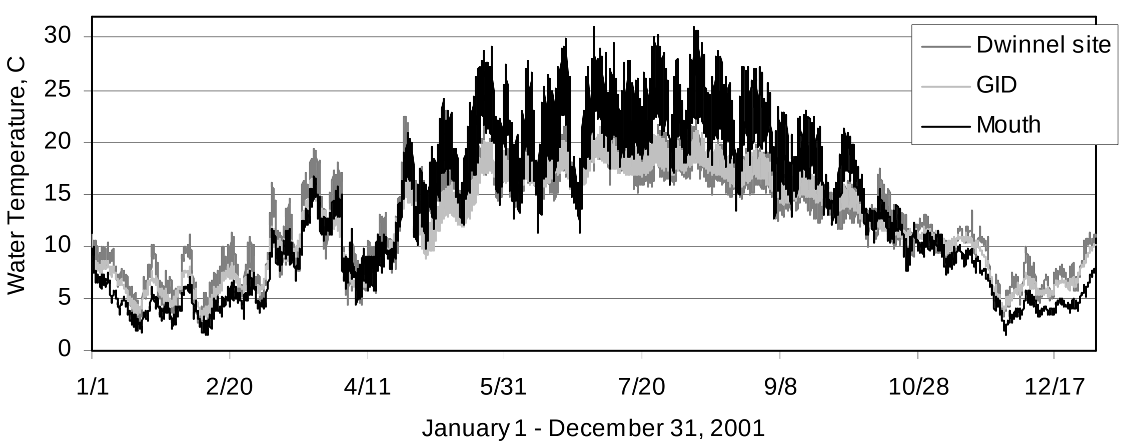


Figure 94. Modeled current conditions water temperature for select Shasta River locations

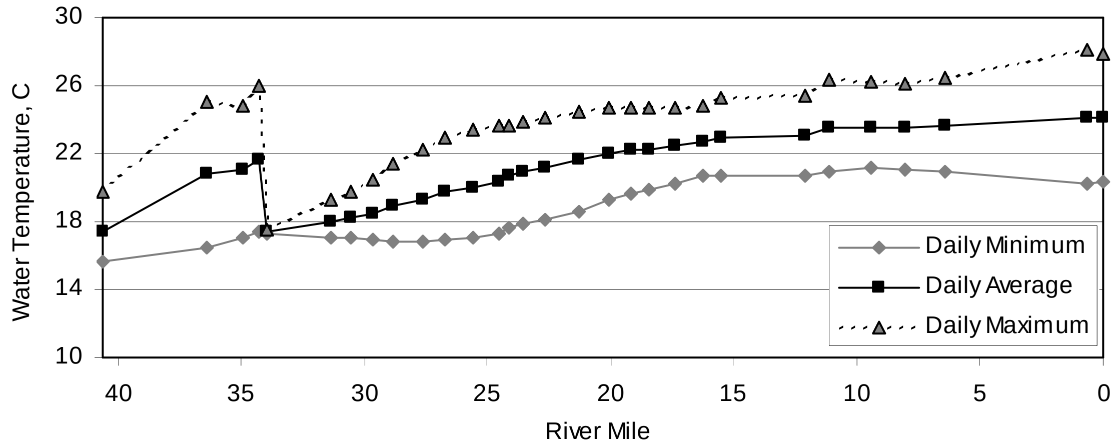


Figure 95. Simulated August 15, 2001 minimum, mean, and maximum water temperature

Minimum Instream Flows

Currently releases from Dwinnell Dam earmarked for water rights that existed before the dam, and the dam may spill during wet years (Vignola and Deas, 2005). Higher minimum instream flows of 10 cfs and 30 cfs from Dwinnell Dam were simulated to explore the effect of increased thermal mass on water temperature. The Shasta River is a small system, thus a yearly release of 30 cfs uses over 21,000 af/yr from Lake Shastina, just under half the available storage. Flow over the length of the Shasta River increased by the volume of the instream flow.

Minimum instream flows mostly reduce thermal variability above Big Springs (RM 33.93) (Figure 96). When the larger Big Springs flow joins the Shasta River, water temperature is very stable, before again being driven by atmospheric heating down the length of the Shasta River. A 10 cfs minimum instream flow has little affect below Big Springs until below the DWR weir (RM 15.52), where maximum water temperature is approximately 0.5°C cooler than without minimum instream flows. A 30 cfs minimum instream flow maintains water temperature below 25°C until approximately river mile 6, in the canyon reach. Decreased diversions from landowners in the upper Shasta River may have a similar affect as instream flow releases from Dwinnell Dam.

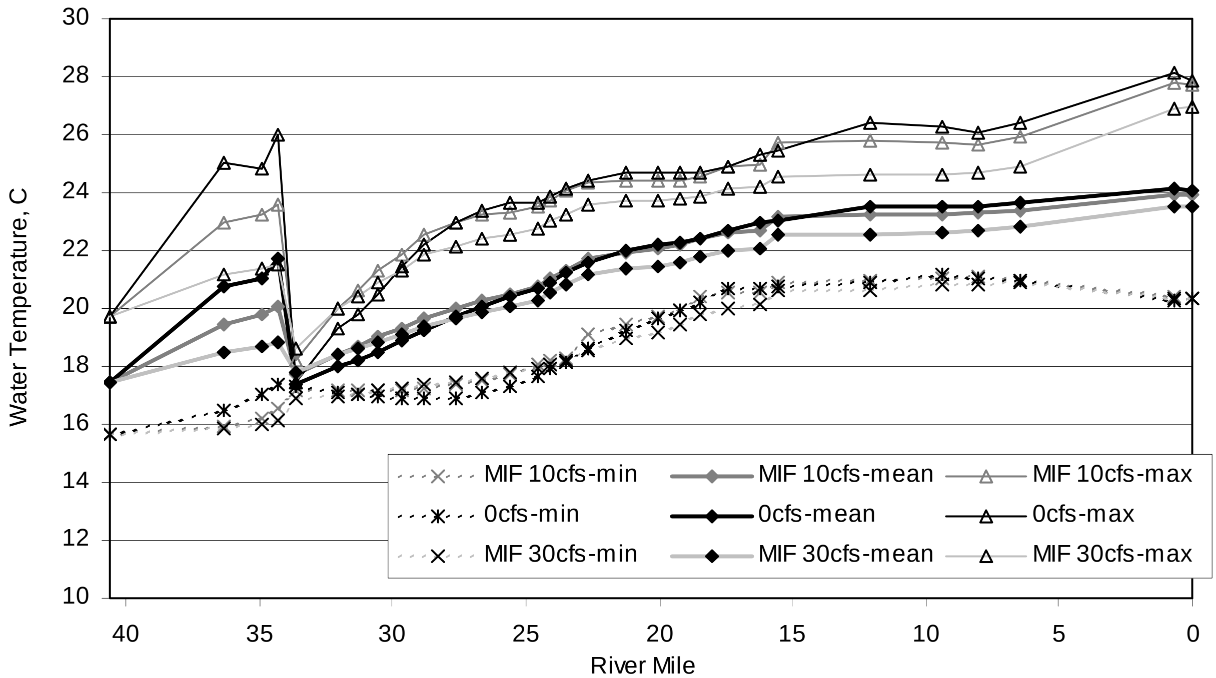


Figure 96. Simulated minimum, mean, and maximum water temperature from minimum instream flow alternatives of 0 cfs, 10 cfs, and 30 cfs, August 15, 2001

Grenada Irrigation District Diversion Alternatives

Currently GID has the most junior water right on the Shasta River, and due to pumping costs, sells its customers the highest priced water in the Shasta basin ($52/af) (TNC, 2005). GID has 2 pumps on the Peters' property at river mile 30.58, each pump has approximately 20 cfs capacity. GID's water right is 40 cfs, although Shasta Valley Watermasters estimate that GID currently pumps approximately 35 cfs because of maintenance on the pumps and junior water right standing (Scott, pers.comm., 2007). Many alterations to GID have been proposed to improve instream habitat, including operating only one pump, no pumping, and moving the diversion point downstream to A12 (RM 24.11) (TNC, 2005).

Flow for these alternatives are straightforward (Figure 97a); however, results for water temperature are more surprising (Figure 97b). Reducing pumping from 35 cfs to 20 cfs during irrigation season has no obvious affect on water temperature. Eliminating pumping completely and dedicating flow for instream uses decreases water temperature by 1°C and 0.8°C, at the DWR weir and mouth, respectively. Moving the GID diversion point to A-12 slightly reduces water temperature over much of the Shasta River. This option slightly increases thermal mass, resulting in a 1°C reduction until the new diversion point. Water remains approximately 0.5°C cooler, although temperature differences diminish longitudinally, with negligible difference by Anderson Road (RM 8.05). This reduction alone is not sufficient to improve instream conditions for native salmon species, although it may be promising when paired with other habitat enhancement options.

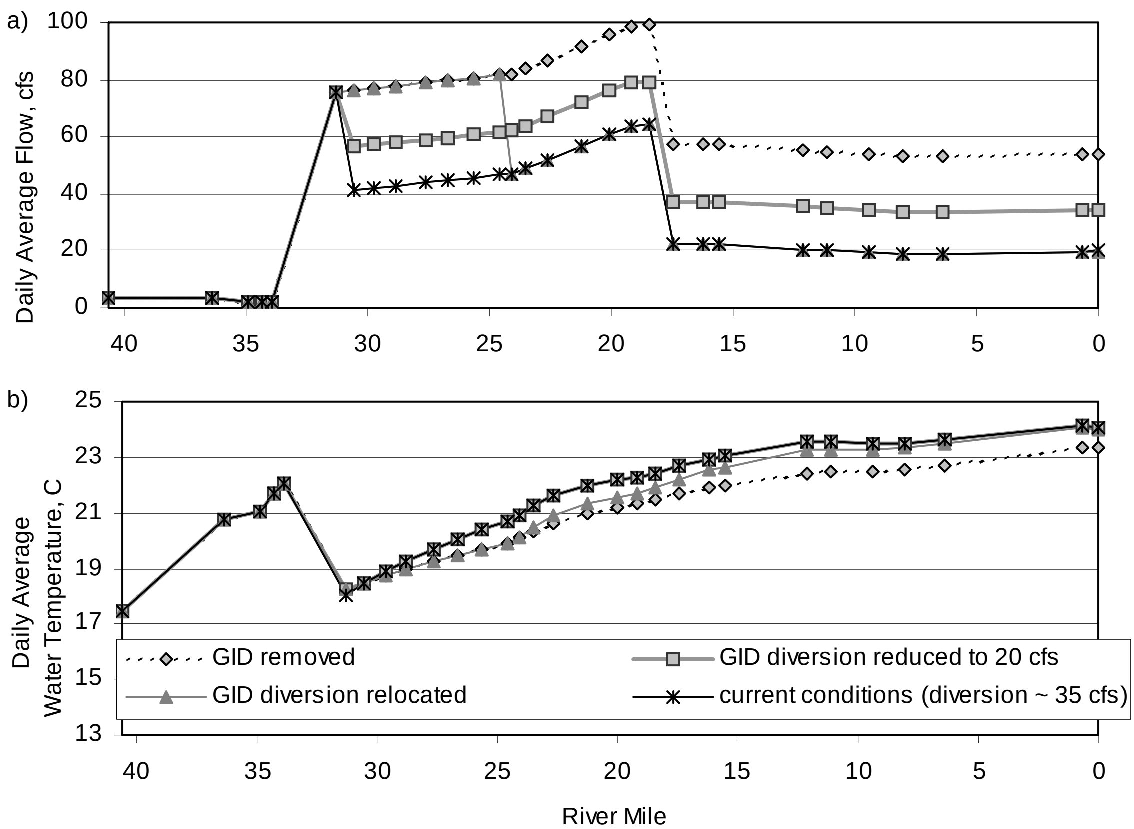


Figure 97. Simulated daily average flow (a), and water temperature (b) of alternatives for Grenada Irrigation District, August 15, 2001

Nelson Ranch Return Flow Analysis

Agricultural tailwater returns to the Shasta River over its length, and is poorly quantified. Flow and water temperature were measured at the Nelson Ranch return flow channel from 7/06 – 10/07 (see fieldwork chapter). This measurement does not account for possible overland or subsurface flow to the Shasta River, but local runoff into the channel is included. During the monitoring period, the Nelson Ranch return flow channel had a baseflow of approximately 0.14 cfs and pulses reached 1.6 cfs. Water depth in the channel is shallow, allowing for rapid heating and cooling driven by atmospheric conditions. Thus the return flow channel has greater thermal variability than the mainstem Shasta River. Minimum recorded water temperature was below 0°C during winter months, and maximum recorded water temperature exceeded 40°C in July and August. Measured flow and water temperature data from 2006-2007 was added as a point-source inflow to the Shasta River at river mile 32.03. Inflow that was added at the Nelson return flow channel was removed from accretions and depletions in the Big Springs to GID reach so that the total water balance remained unchanged.

There was little change to water temperature when measured return flow (Nelson RF run) was compared with the current conditions run (Figure 98). Measured return flow volume was increased by 5 cfs and 10 cfs, using the same water temperature boundary condition, to explore how much tailwater can be returned to the Shasta River before it affects temperature. Each additional 5 cfs of return flow increased water temperature by 1°C in the Shasta river at the tailwater junction, and temperature differences lasted until river mile 24.57 (just upstream of A-12). There are only small differences for minimum daily temperature. During winter, return flow is a cold-water input to the Shasta River, although temperature is not limiting.

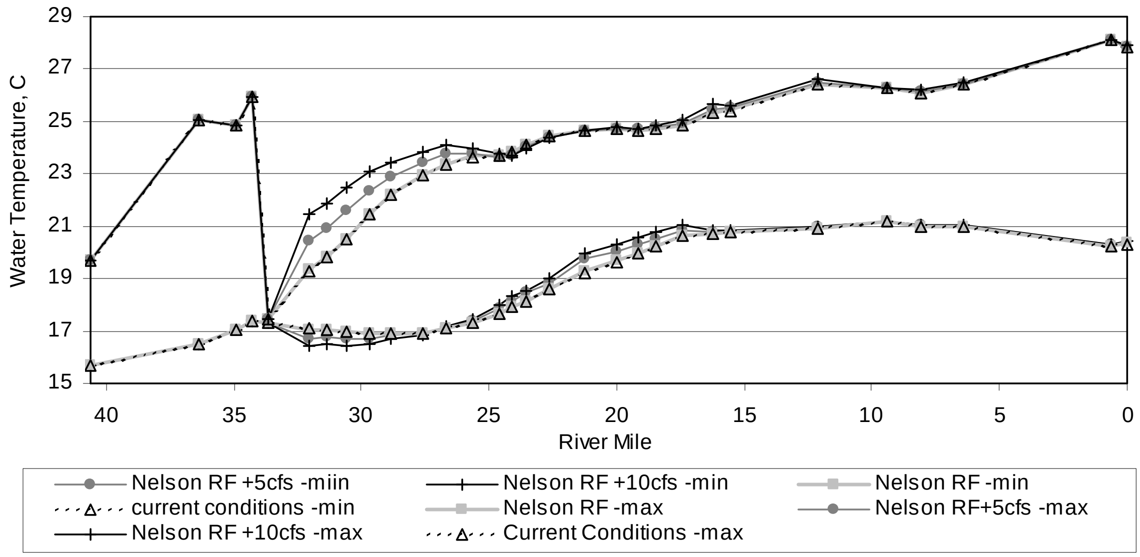


Figure 98. Simulated minimum and maximum daily water temperature for Nelson Ranch return flow alternatives, August 15, 2001

Riparian Vegetation Alternatives

Aside from springflows, meteorological conditions mostly drive thermal conditions in the Shasta River. Water temperature response to solar radiation varies seasonally with maximum loading occurring during summer months when days are long, there is little rain, and few cloudy days. Riparian shading can reduce or largely eliminate thermal heating from solar radiation.

Riparian shading input data includes tree height and percentage of light transmitted through the canopy (not reflected or absorbed). Current conditions assumes 22 ft (6.7 m) trees on both banks of the Shasta River with variable light transmittance densities reflecting sparse vegetation (Figure 74). A run with 1 ft (0.3 m) tree height (representing bank height) was used to predict water temperature without shading, as an upper bound for water temperature. Fully vegetated riparian canopies transmit approximately 20% of available light to the river (Lowney, 2000). Thus, maximum shade with 35 ft (10.7 m) trees and 20% transmittance was modeled as a water temperature lower bound; although a full gallery forest may not be attainable on the Shasta River because of anoxic soils (Webb, pers. comm., 2006).

Two additional runs were completed to represent water temperature response to forest succession. The first run simulated growth of bulrush and Tule reeds in the channel, which could be expected if the river were completely fenced (Abbott, 2002). Where transmittance exceeded 85% under current shading conditions, 5 ft (1.5 m) trees were added (representing reeds and bulrush) and transmittance was given a value of 85%. Next, the 5 ft (1.5 m) trees were increased to 15 ft (4.6 m), with 50% transmittance to represent an immature forest (NCRWQCB, 2006).

Model runs suggest that Shasta River water temperature is sensitive to riparian shading (Figure 99). Temperature differences between the upper and lower bounds of riparian vegetation increase with downstream distance from Big Springs because atmospheric heating is reduced (RM 33.93). At the mouth, average daily water temperature is nearly 2.5°C cooler under the full shade alternative than the 1 ft (0.3 m) shade alternative, and differences between the two runs for the daily maximum and minimum water temperature on August 15th at the mouth are 2.8°C and 2.3°C, respectively. The biggest improvement to water temperature occurs in the upper Shasta River, between Dwinnell Dam and Big Springs, where daily average water temperature below 19°C may be possible with a fully vegetated stream bank. That reach was historically used for coho spawning, and continues to be used, as coho redds were discovered in this reach in fall 2007 (Jeffres, pers.comm., 2007). Most temperature improvements occur with full reforestation, and the 5 ft (1.5 m) and 15 ft (4.6 m) tree model runs had little effect on water temperature.

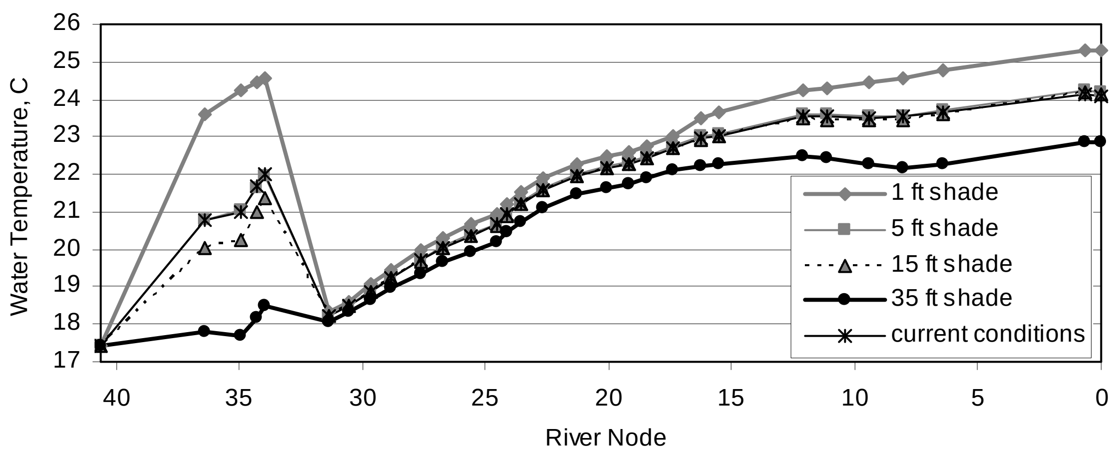


Figure 99. Simulated mean daily water temperature with riparian shading, August 15, 2001

Fully Restore Big Springs

A unique attribute of the Shasta River is the thermal stability of substantial springflows, which may enter the river either notably warmer, nearly the same, or considerably cooler than mainstem water temperatures depending on the time of year. In general, groundwater-dominated river systems, like the Shasta River, have a more stable flow and thermal regime than those dominated by surface water (Sear et al., 1999). Big Springs contributes most of the spring-derived water, and thus is likely critical to restoration efforts on the Shasta River.

Shasta River conditions were simulated for a fully restored Big Springs complex, which assumes minimum summer flow of 104 cfs and maximum winter flow of 124 cfs (current flows vary between 70-90 cfs), and consistently cool temperatures between 10.4 – 12.5°C (current summer water temperature exceeds 20°C at the Shasta River at Big

Springs). To attain these conditions, flow from Big Springs would have to be shaded, channelized and flow directly to the Shasta River.

Modeling suggests that flow at the mouth could be expected to increase 30-50 cfs during all seasons if Big Springs were fully restored (Figure 100). Water temperature is greatly reduced because large contributions of cool water are added, and increased flow increases thermal mass to maintain cool conditions over the length of the Shasta River (Figure 101). Maximum water temperature still exceeds 25°C during summer at the mouth, although it never exceeds 30°C, as occurs under current conditions.

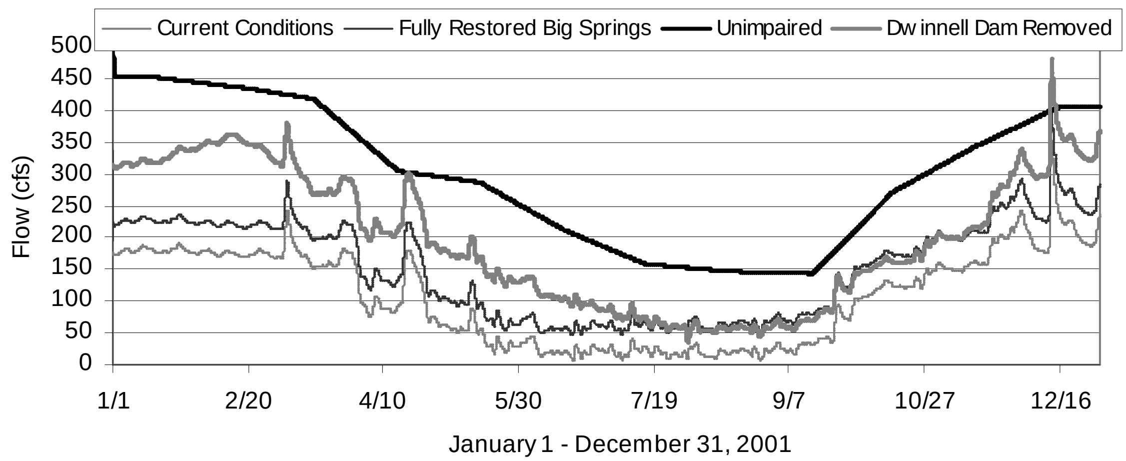


Figure 100. Simulated flow at the mouth under unimpaired, restored Big Springs complex, and current conditions

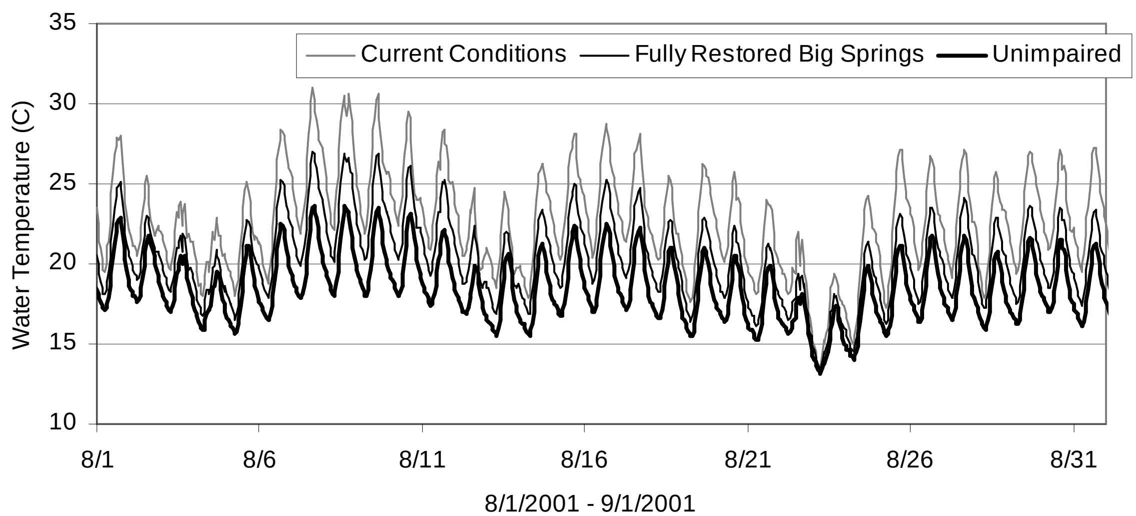


Figure 101. Simulated water temperature at the mouth under unimpaired, restored Big Springs complex, and current conditions

Thermal improvements are more pronounced upstream. Water temperature at GID rarely exceeds 16°C, providing optimal habitat for salmon (Figure 102). However, property along Big Springs Creek is privately owned, and water is appropriated to water right holders. Big Springs Irrigation District pumps groundwater, which has not been adjudicated in the Shasta Basin (NRC, 2004). Therefore, completely restoring Big Springs is legally difficult.

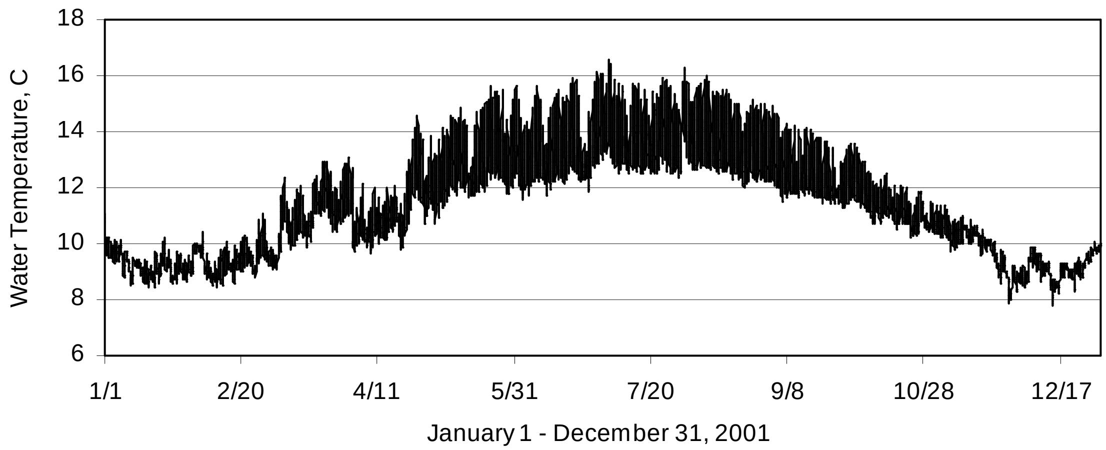


Figure 102. Simulated water temperature at GID under fully restored Big Springs conditions

Remove Dwinnell Dam

Lake Shastina was impounded in 1928. The 1923 water right allowed 60,000 af to be stored from October to June, although maximum operating capacity is approximately 50,000 af (Booher et al., 1960s). The reservoir has substantial seepage losses through underlying volcaniclastic rocks (Vignola and Deas, 2005). Direct reservoir outflow includes seepage, minimal controlled releases of up to 10 cfs (0.28 cms), and relatively infrequent uncontrolled winter spill events (e.g. 1964 and 1997) (Vignola and Deas 2005; Crabill, pers.comm., 2007). Removing Dwinnell Dam has been recommended because the dam is aging, highly inefficient, blocks access to 22% of upstream coho habitat, and degrades downstream habitat by trapping spawning gravels, inhibiting geomorphically important peak flows, and loading nutrients which may impair downstream water quality (NRC, 2004).

Estimated unimpaired flow and water temperature below Dwinnell Dam were used to model instream conditions without Dwinnell Dam. For the boundary condition at Dwinnell Dam, this assumes that upstream tributaries have also been fully restored. Currently, water temperatures at Edgewood in summer are quite warm, and without restoration of tributaries, atmospheric heating from the headwaters to the damsite may not yield the cooler water that is assumed here.

Currently 15,000 af is diverted from Parks Creek to Dwinnell Dam each year (Vignola and Deas, 2005). For this run, flow from Parks Creek were increased by 20.72 cfs per day (15,000 af/yr), except when this raised Parks Creek above unimpaired flow levels from 7/4 – 9/22 (Figure 103). Since flow was constrained to unimpaired levels during summer, only 13,755 af of water was added to Parks Creek throughout the year. Water temperature from Parks Creek was left unchanged from current conditions. In reality, greater thermal mass would reduce atmospheric heating and decrease water temperature, especially during spring runoff.

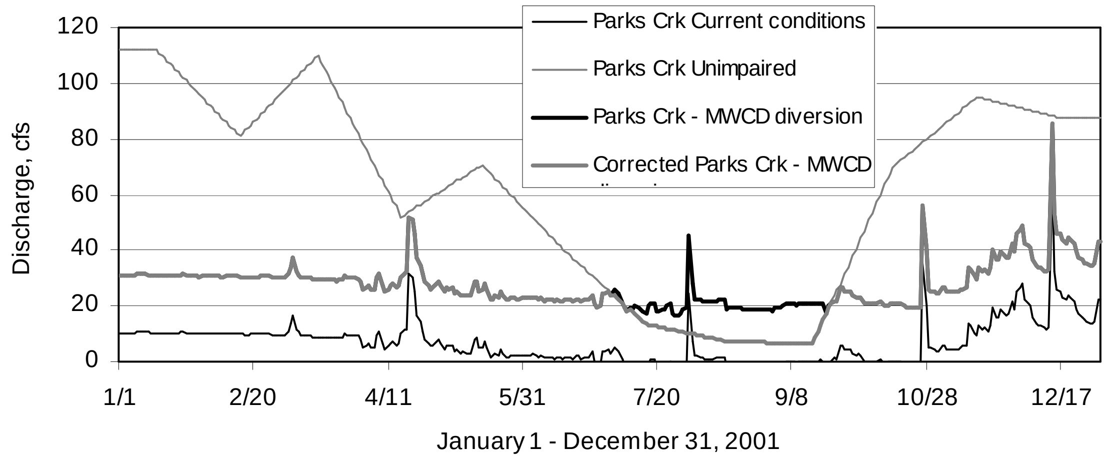


Figure 103. Parks Creek boundary conditions

Results indicate that from mid-July to mid-November, removing Dwinnell Dam or restoring Big Springs leads to similar flow conditions at the mouth (Figure 100), assuming full restoration of tributaries above Dwinnell. In other months, flow is at least 100 cfs greater than restoring Big Springs, and up to 150 cfs greater than current conditions. Removing Dwinnell Dam reduces water temperature throughout the length of the Shasta River, especially during spring, although the reduction is most pronounced above the Big Springs complex, the historic rearing habitat of coho salmon (Jeffres et al., 2008) (Figure 104, Figure 105). During spring and fall, mean water temperature from the upper Shasta River is below equilibrium temperature, and remains slightly below equilibrium temperature even below Big Springs, causing the river to warm longitudinally (Figure 104). Mean summer water temperature typically remains below 18°C until GID (Figure 105). Removing Dwinnell Dam maintains cool temperatures through June in much of the river, so that summer and early fall are the only critical months for water temperature (critically warm temperature begin by mid-May under current conditions) (Figure 106).

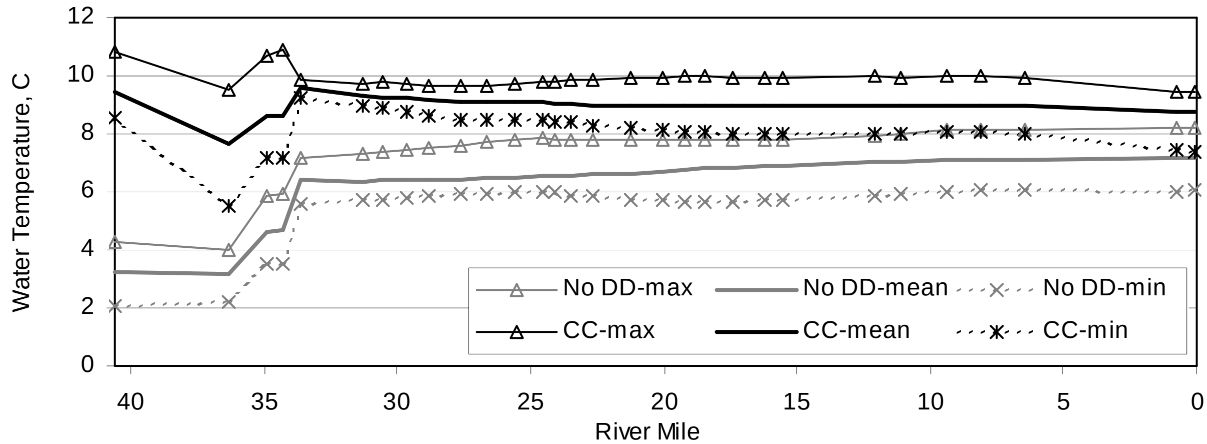


Figure 104. Simulated minimum, mean, and maximum water temperature with Dwinnell Dam removed (No DD) and current conditions (CC), March 15, 2001

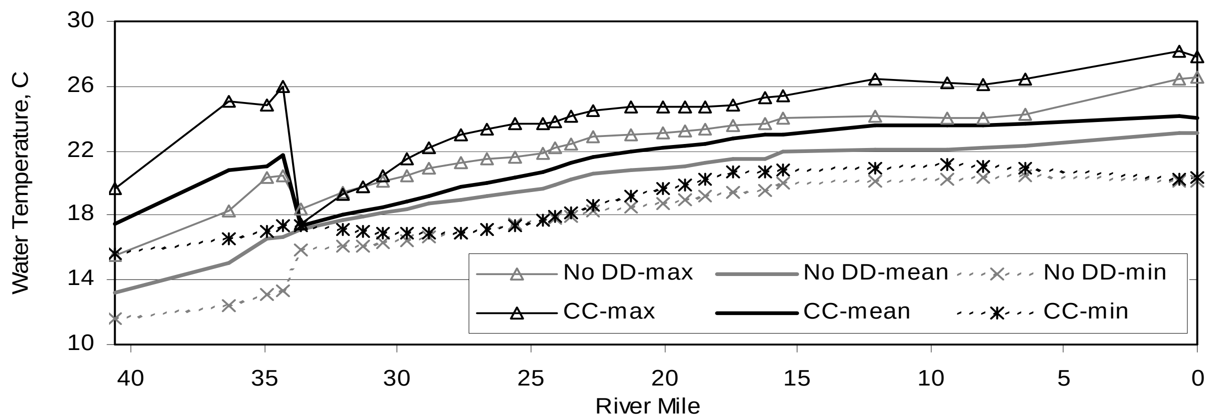


Figure 105. Simulated minimum, mean, and maximum water temperature with Dwinnell Dam removed (No DD) and current conditions (CC), August 15, 2001

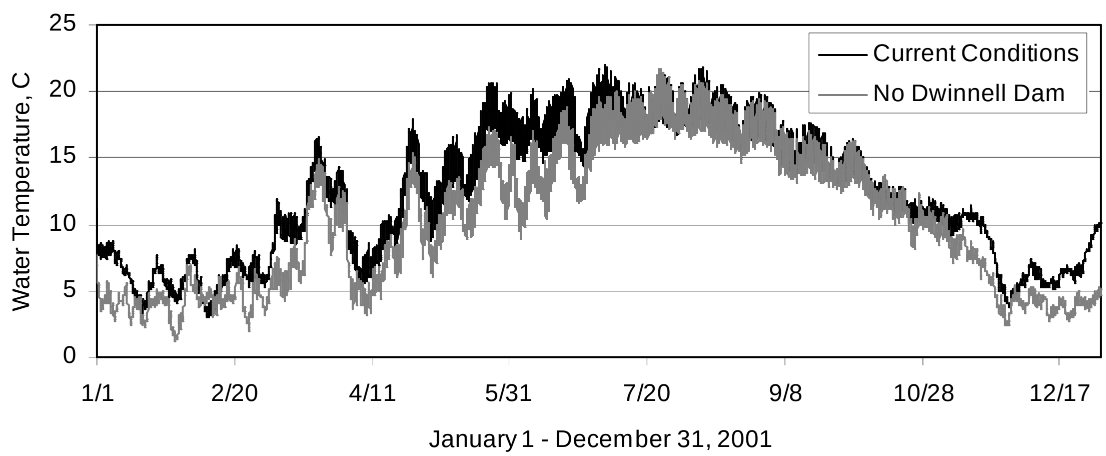


Figure 106. Water temperature at GID with current conditions and without Dwinnell Dam

Comparison of Results

Estimated hydrologic and thermal conditions of the Shasta River under the restoration alternatives discussed above are compared in the following paragraphs. The river is divided into four reaches, Dwinnell Dam to Big Springs Creek, Big Springs Creek to the SWUA diversion, the SWUA diversion to Yreka Creek, and Yreka Creek to the Mouth for comparison of flow and water temperature results.

In general, the Shasta River below Dwinnell Dam is thermally-limited, meaning under current conditions, water temperature (exacerbated by low instream flows) is the primary factor inhibiting salmon survival (NRC, 2004). Overall, options exist to cool the Shasta River. Initial cool water conditions in the first or second reaches are necessary for restoration alternatives to be effective because it is easier to maintain cold water than cool warm water. Additionally, a mix of alternatives, each collectively improving conditions, is most helpful to enhance instream conditions for native salmon species. Different restoration alternatives result in improvements to different reaches or seasons; thus, it is important to work with fish biologists and local stakeholders to determine the spatial and temporal needs of salmon species to ensure their survival. Habitat quality considerations other than instream flow and water temperature are ignored here.

Below Dwinnell Dam to Big Springs Creek

Historically, the reach from Dwinnell Dam (RM 40.62) to Big Springs Creek (RM 33.67) provided coho spawning and rearing habitat, although high water temperatures and accessibility problems now inhibit coho rearing throughout summer (and additional coho rearing habitat above Dwinnell Dam is no longer accessible). The current conditions, no GID diversion, and restoring Big Springs alternatives result in extreme low flow conditions (there is no difference between these alternatives in this reach) (Figure 107, Figure 108). Minimum instream flow releases of 30 cfs, removing Dwinnell Dam, and unimpaired conditions all allow moderate instream flow. Removing Dwinnell Dam and unimpaired conditions are unique because dry summer conditions last for a shorter period than all other restoration alternatives. With unimpaired conditions, weekly mean flow below 30 cfs persists only from August through October, and flow below 50 cfs lasts from July through mid-November (Figure 107). Thus, to improve instream flow conditions below Dwinnell Dam and above Big Springs, releases from Dwinnell Dam or removing the dam are the best options to increase flow, with the dam removal option resulting in greater seasonal effects on flow conditions. Increasing flow increases the area of habitat available, and reduces water temperature by increasing thermal mass.

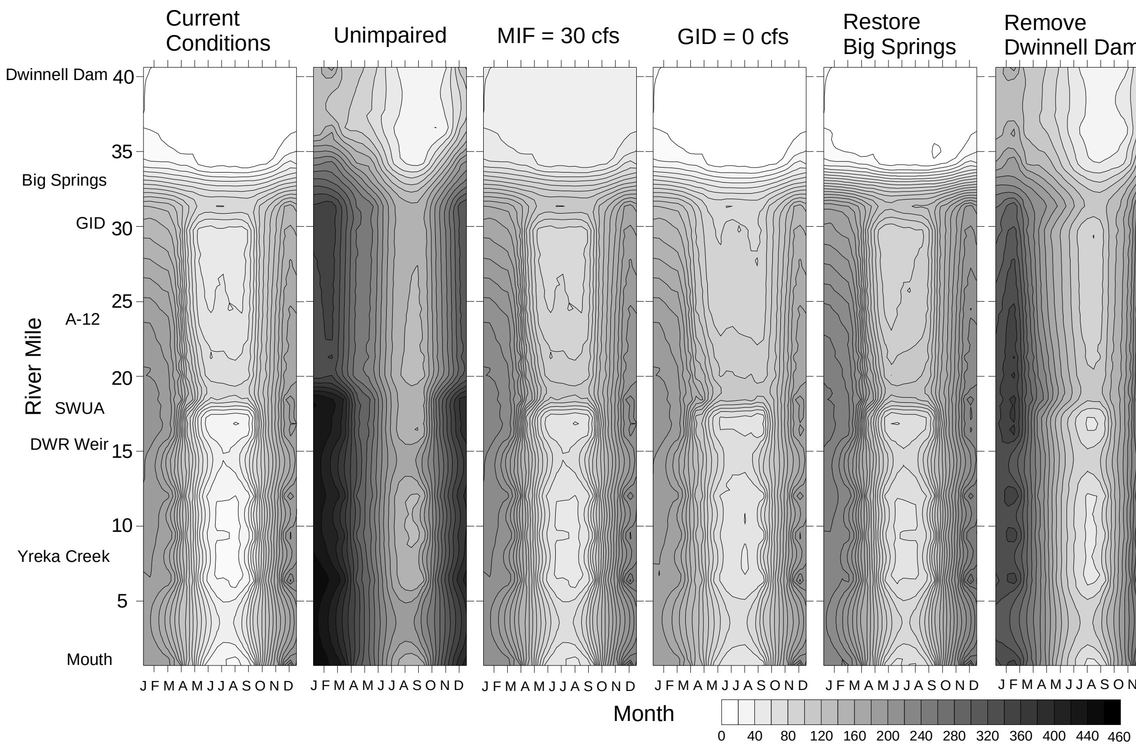


Figure 107. Simulated spatial and temporal weekly mean flow (cfs)

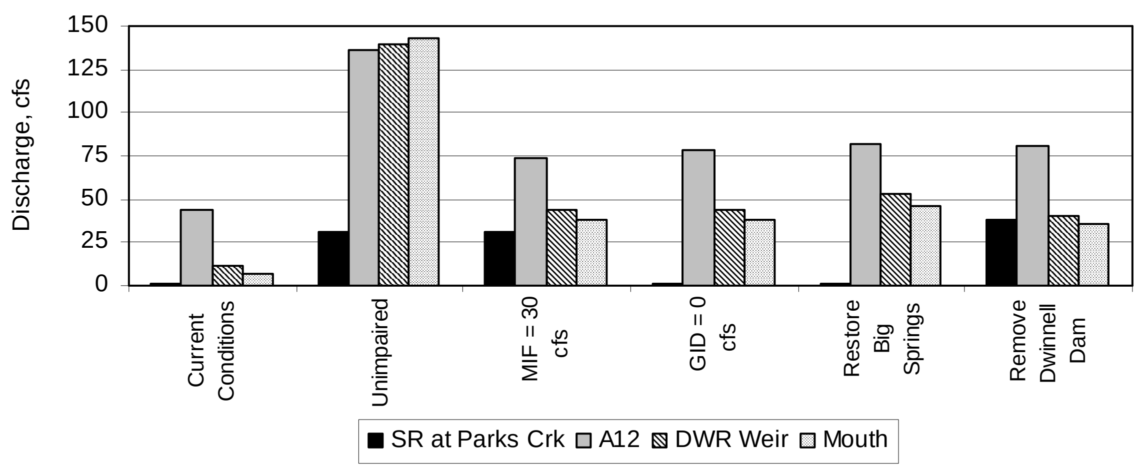


Figure 108. Simulated minimum annual hourly flow for restoration alternatives

The current conditions, no GID diversion, and restore Big Springs alternatives result in maximum weekly mean thermal conditions greater than 22°C (Figure 109), and a maximum hourly water temperature of 28°C on August 8, 2001 (there is no difference between these runs in the Dwinnell Dam to Big Springs reach) (Figure 110). Minimum instream flow releases of 30 cfs reduce maximum weekly mean water temperature to just above 20.2°C because greater thermal mass reduces atmospheric heating. Riparian shading of 35 feet on both river banks reduces maximum weekly mean water temperature slightly more, to 19.9°C, because solar radiation is partially blocked. Removing Dwinnell

Dam and unimpaired conditions reduce temperatures the most, with maximum weekly mean temperature 19.6°C and 18.5°C, respectively. Like flow results, these two alternatives have a much shorter warm season than all other alternatives (Figure 109).

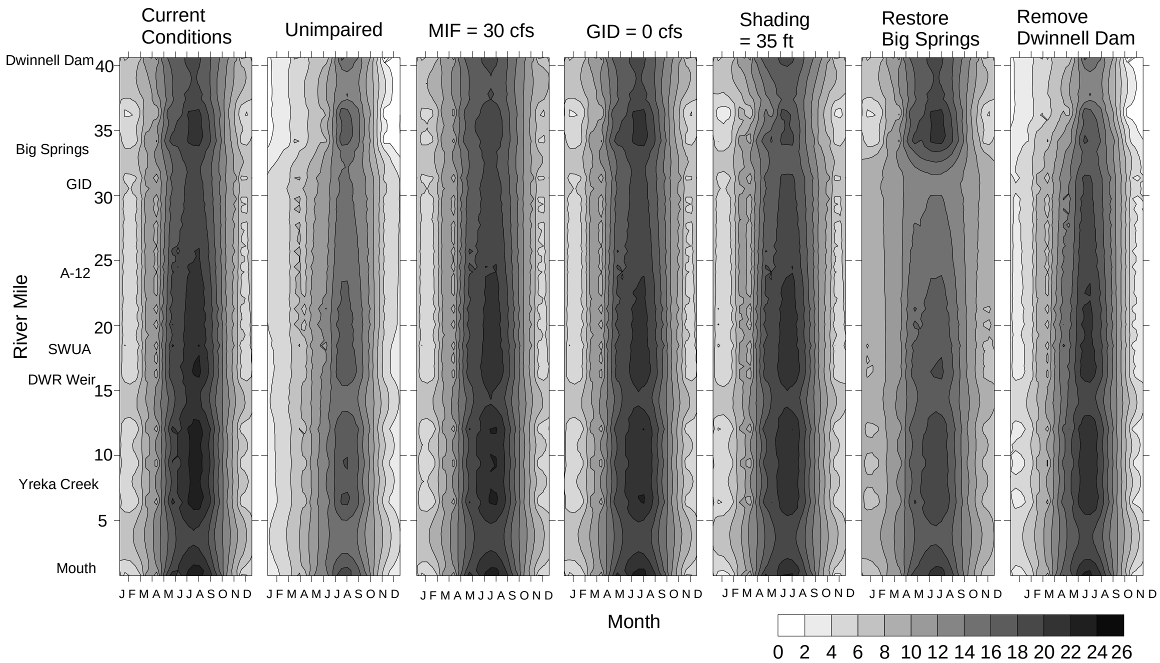


Figure 109. Simulated spatial and temporal mean weekly water temperature (C)

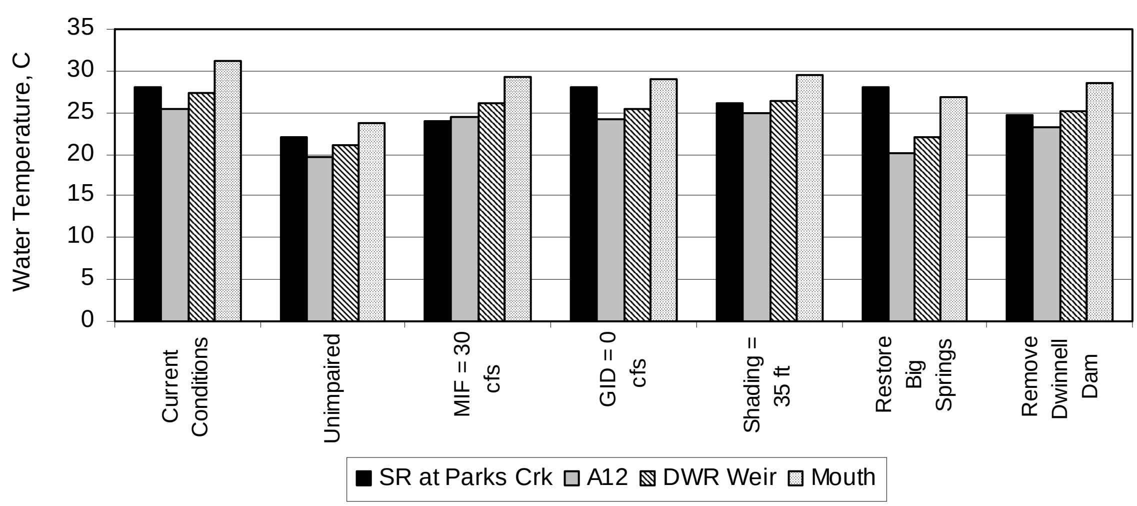


Figure 110. Simulated maximum annual hourly water temperature for restoration alternatives

Big Springs Creek to SWUA Diversion

The reach from Big Springs Creek (RM 33.67) to the SWUA diversion (RM 17.85) includes the upper portion of the alluvial Shasta Valley. Some spawning and rearing occur high in this reach (Jeffres et al., 2008). Big Springs Creek contributes flow to the Shasta River, and GID and SWUA are large diversions, making these three locations transition zones for instream flow (Figure 107). Current river conditions have the lowest instream flow of all alternatives analyzed for this reach (Figure 107, Figure 108). Minimum instream flows, elimination of GID diversions, and restoring Big Springs Creek all lead to similar flow results, with flow increased by 30-40 cfs. Removing Dwinnell Dam is similar to Restoring Big Springs in terms of yearly minimum flow conditions (Figure 108), although like the previous reach, low flow conditions are short-lived when Dwinnell Dam is removed (Figure 107). Unimpaired conditions show a marked increase in flow from restoring Big Springs or removing Dwinnell Dam because all diversions are also eliminated.

Atmospheric heating increases water temperature longitudinally in all alternatives analyzed, which is consistent with field monitoring findings on the Nelson Ranch (RM 27.36 – 32.10) (Null, field monitoring chapter, 2008). For discussion of thermal conditions, Highway A-12 divides this reach into upper and lower sub-reaches. Current conditions has the highest water temperatures of all alternatives (Figure 109, Figure 110, Table 18). Minimum instream flows, 35 ft (10.7 m) riparian shading, no diversion at GID, and removing Dwinnell Dam have only minimal thermal improvements, indicating cool water is needed in the upper portions of the Shasta River or its tributaries for management alternatives to be beneficial. Under the restored Big Springs Creek and unimpaired alternatives, Big Springs Creek has a stable year-round thermal regime, resulting in appreciable improvements to water temperature throughout the Big Springs to SWUA reach. Additionally, winter water temperature is warmer because springflow contributions are warmer than equilibrium river temperature during winter months, resulting in improved winter conditions for native salmon (Moyle, 2002).

Table 18. Maximum weekly mean water temperature (C) in the upper and lower Big Springs to SWUA reach

	Big Springs – A-12 (RM 33.67 – 24.11)	A-12 – SWUA Diversion (RM 24.11 – 17.85)
Current Conditions	21.3	23.0
MIF = 30 cfs	20.9	22.4
Shading = 35 ft (10.7 m)	20.8	22.5
GID = 0	20.6	22.0
Remove Dwinnell Dam	20.5	21.8
Restore Big Springs	16.2	18.5
Unimpaired	16.2	17.9

SWUA Diversion to Yreka Creek

The reach between the SWUA diversion (RM 17.85) and Yreka Creek (RM 7.88) is in the downstream portion of the alluvial Shasta River Valley. Most large diversions occur upstream, resulting in extreme low flow conditions in this reach during summer. Atmospheric heating exacerbated by low flows, heats the river longitudinally. Minimum annual flow and maximum annual water temperature are presented at the DWR weir (RM 15.52) near the upstream end of this reach (Figure 108, Figure 110). Modeled weekly mean flow under current conditions has a minimum of 15 cfs (Figure 107). Minimum instream flows, elimination of GID diversions, restoring Big Springs, and removing

Dwinnell Dam all maintain weekly mean flow of at least 44.8 – 56.6 cfs. As expected, flow is substantially higher, 132.7 cfs, with unimpaired conditions, and higher flow conditions extend through winter. Higher winter flows also occur to a lesser extent when Big Springs is restored or Dwinnell Dam removed.

Current conditions have a maximum weekly mean water temperature of 24.2°C (Figure 109) at the DWR weir. All restoration alternatives cause only slight thermal improvements, with the exception of restoring Big Springs and unimpaired conditions (Figure 109, Figure 110), showing that the thermal benefits of restoring Big Springs extend downstream for much of the Shasta River, and restoring Big Springs may be imperative to enhancing conditions in downstream reaches. Unimpaired thermal conditions are better than restoring Big Springs alone, suggesting a mix of restoration alternatives would further improve instream conditions.

Yreka Creek to the Mouth

Recorded and simulated water temperatures are consistently highest at the mouth (RM 0.72). All anadromous fish must migrate though the mouth of the Shasta River, although coho, Chinook, and steelhead out-migrate by mid-July (with the possible exception of Type II or Type III Chinook salmon), and spawners generally do not enter the Shasta River until mid-September, largely avoiding months with the warmest thermal conditions (CDFG, 1997; NCRWQCB, 2006). Flow conditions in this reach are similar to the previous reach, modeled minimum weekly average flow is 14 cfs with current conditions (Figure 107), and hourly flow reached an annual minimum of 6.5 cfs on June 20, 2001 (Figure 108). To increase flow at the mouth of the Shasta River, minimum instream flows, decreased diversions, restoring flow from Big Springs, or removing Dwinnell Dam are the most promising alternatives, raising flow at the mouth to a minimum weekly mean of 43.6 – 55.4 cfs (Figure 107, Figure 108). Yreka Creek increases flow slightly during summer conditions with unimpaired conditions.

Water temperature in this reach is also similar to the previous reach, except temperatures are slightly warmer due to continued atmospheric heating. Increasing riparian vegetation raised maximum weekly mean water temperature 0.4°C between the two reaches, the smallest change of all model runs; while maximum weekly mean temperature increased 1.8°C with the restored Big Springs run, the most change of all runs. This implies water temperature at the mouth is farther from equilibrium conditions under the restored Big Springs alternative.

Welsh et al. (2001) found that coho were present in tributaries to California's Mattole River when maximum weekly average water temperature (MWAT) was 16.7°C or less. Water temperature estimates may be too high because MWAT for a representative summer week (8/5/01 – 8/11/01) in model runs outlined here show unimpaired water temperature warmer than Welsh's ideal MWAT of 16.7°C for much of the Shasta River (Figure 111). It is also possible that historic MWAT water temperature was greater than 16.7°C in the Shasta River, but abundant food kept fish productivity high; or fish migrated through the lower part of the Shasta River during spring or fall under cooler thermal conditions. Model results suggest removing Dwinnell Dam would provide adequate summer thermal conditions for coho for approximately five river miles immediately downstream of the damsite, and unimpaired and restoring Big Springs Creek would provide approximately ten miles of optimal thermal conditions directly downstream of the

Big Springs confluence. Water temperature remains above the 16.7°C target with all other alternatives.

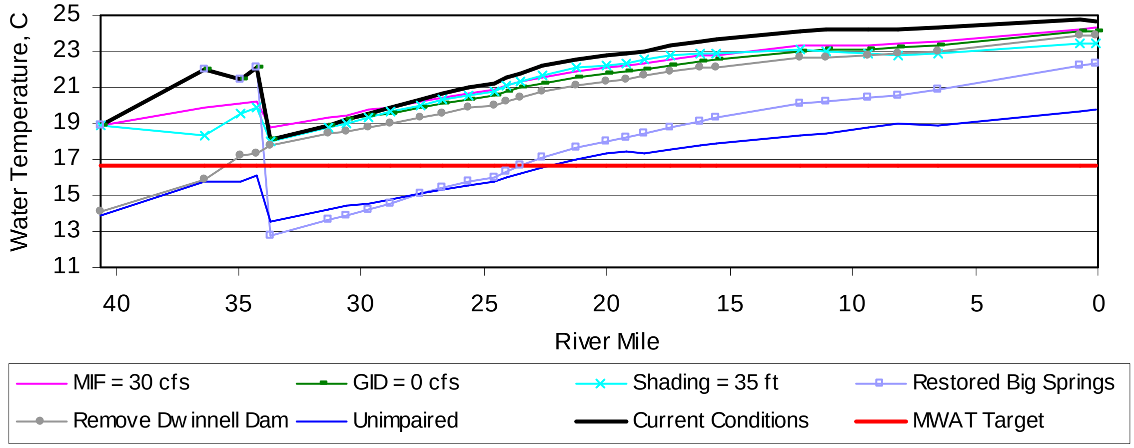


Figure 111. Longitudinal maximum weekly average water temperature (MWAT) under different restoration alternatives with MWAT target, 8/5/01 – 8/11/01

Limitations

Simulation modeling is a good approach for representing Shasta River conditions and evaluating potential changes to instream conditions using different management alternatives. Limitations exist, primarily from simplifications inherent in modeling studies, lack of input data at critical locations along the river, and tributaries modeled as boundary conditions rather than discrete tributaries.

Representation of the Shasta River has been simplified, as occurs with all modeling studies. The river from Dwinnell Dam to the mouth at the Klamath River has been represented with 999 nodes, the maximum allowable in RMS. Channel geometry could be improved, particularly with physical measurements of surface width, depth, bank height, and cross-sectional geometry at additional sites along the Shasta River. Additionally, the modeled channel has steep banks which may not accurately represent flood flows.

Small diversions, tailwater returns, and most groundwater flow (percolation, infiltration, subsurface discharge, small springs and seeps) are lumped and modeled as accretions or depletions on a reach-scale. Quantifying groundwater flow and temperature, including interactions between Lake Shastina and the Shasta River, and stability of groundwater contributions, would help improve understanding of the Shasta River, particularly since the Shasta River is heavily influenced by groundwater. Similarly further studies on tailwater contributions, such as timing, quantity, and thermal variability would help to aid understanding of the cumulative effects of tailwater returns to the Shasta River, improving modeling efforts and aiding management decisions.

Collecting discharge and water temperature data below Dwinnell Dam and at major tributaries to the Shasta River would greatly improve simulation results and advance understanding of the river. Parks Creek, Big Springs Creek, the Little Shasta River, Oregon Slough, and Yreka Creek are tributaries for which additional data would be most useful. Long-term (multiple years) discharge and water temperature at these locations would improve simulation results and would allow additional years to be modeled accurately. Tributaries could be modeled explicitly if measured flow and water temperature data existed, instead of modeled as boundary conditions to the Shasta River. This is critical for understanding the seasonal, daily, and hourly role of tributaries on Shasta River flow and temperature conditions. Increased understanding of the quantity of tributary water entering the Shasta River at all times, how it affects transit time of the river, as well the thermal characteristics of these inflows is crucial to understanding current Shasta River conditions, and will help focus management decisions.

Discussion

Historically, Shasta River flow was derived from spring inflows, which provided persistent baseflow at consistent year-round water temperatures. Flow was enhanced with rain and snow runoff from tributaries such as Parks Creek, Carrick Creek, and the Little Shasta River. Water temperature was influenced by the thermal regime of headwaters and tributaries, and springwater inflow was typically warmer than equilibrium river temperature during winter and cooler during summer, providing ideal instream conditions for native salmon. Atmospheric heating, primarily from solar radiation and air temperature, also was a major influence on river temperature, heating as distance from headwaters or tributaries increased. Flows have been altered and diminished by construction of Dwinnell Dam, surface water diversions, and groundwater pumping, resulting in low instream flows in the Shasta River; while water temperature has increased from diversion of cool spring water, tailwater return flow, low flow conditions, and reduction of riparian shading.

Although input data could be improved, this analysis largely constrains the problem to provide a reasonable estimate of current and potential flows and temperatures for a representative year in the Shasta River. A range of alternatives was analyzed, targeting increasing flow conditions, reducing water temperature, or combinations of the two. Overall, results suggest that restoration alternatives exist to increase flow and cool the Shasta River; and that a mix of restoration approaches is likely needed. Cool water conditions in the upper reaches of the Shasta River (below Dwinnell Dam) is required for restoration options to be effective because maintaining water temperature is more feasible than cooling a thermally loaded river. Given cool water in the upper reach, restoration options such as improving riparian shading or increasing flow (from reduced diversions, minimum instream flows, or dam removal) best maintain conditions longitudinally.

This study also indicates that substituting higher quality water can sometimes benefit native species without increasing environmental water allocations. In this analysis, the Shasta River heated longitudinally at a similar rate when additional riparian shading reduced solar radiation as when thermal mass was increased by raising flow. This finding has important implications for environmental water use efficiency because restoration decisions that target preserving cool water sources and maintaining temperatures are as effective as increasing instream flow, where water temperature is limiting. As this case study shows, a mix of restoration alternatives targeting both instream flow and water quality should be analyzed for improvements to instream habitat (in addition to determining minimum flow levels for fish to bypass physical barriers). In this way, water use efficiency is considered for environmental water uses, in addition to urban and agricultural water uses.

The following points summarize the key findings for enhancing instream flow and thermal conditions in the Shasta River:

- Cool water is needed in the upper reaches of the Shasta River (below Dwinnell Dam or near the Big Springs Complex). Restoration and management options can then effectively maintain temperature. Without cool water, most management alternatives become largely ineffective.

- A mix of restoration strategies provides the greatest improvements to instream habitat.

- Restoration alternatives benefit different river reaches. Managers should work with fish biologists to target reaches that have the most benefit for desired fish species.

- The effects of restoring Big Springs Creek extend downstream through much of the Shasta River, making this a promising option for improving instream flow and temperature conditions.

- Each additional 5 cfs of modeled return flow (at temperatures measured from the Nelson Ranch return flow during summer 2006) increase water temperature in the Shasta River on Nelson Ranch by 1ºC.

- Simulation modeling is a good technique for highlighting promising restoration alternatives and eliminating alternatives that may not provide the hoped for benefits.
    - Additional simulation can refine promising restoration alternatives.

- Improving water quality can sometimes reduce instream flow needs for native salmon.
    - Well defined objectives help water managers improve instream flow conditions for native salmon without unnecessarily elevated instream flow levels.

References

Abbott, A.G.P. 2002. The Effect of Riparian Vegetation on Stream Temperature in the Shasta River. Master's thesis. U.C. Davis Dept. of Civil and Environmental Engineering.

Booher, L.J., C.E., Houston, A.D. Reed, M.V. Maxwell, S.D. Nelson. 1960s. Analysis of the Montague Water Conservation District. University of California Agricultural Extension Service. Yreka, CA.

CDFG (CA. Dept. of Fish and Game). 1997. A biological needs assessment for anadromous fish in the Shasta River, Siskiyou County, California. CA. Dept. of Fish and Game, Northern California-North Coast Region. Redding, CA.

CDWR (CA Dept. of Water Resources). 1930-1990 Reports. Shasta River Watermaster Service Area Reports.

CDWR (CA Dept. of Water Resources). 1998. *Klamath River Basin Natural Flow*. Office Memo, March 13.

CDWR (CA Dept. of Water Resources). 2001. Watershed Study Plan: Water Quality and Aquatic Habitat Characterization in the Shasta River. Prepared by G.L. Boles. Northern District. Red Bluff, CA.

CDWR (CA Dept. of Water Resources) Watermaster Report. 2006. Lower Shasta River, Upper Shasta River, and Little Shasta River service areas. Available online (http://www.nd.water.ca.gov/PPAs/Watermasters/ServiceAreas/ShastaRiver/index.cfm). Accessed 4/2007.

CDEC (CA Dept. of Water Resources – California Data Exchange Center). Available online (http://cdec.water.ca.gov/). Accessed 4/2007.

Chapra, S.C. 1997. Surface Water-Quality Modeling. McGraw-Hill. NY, NY.

Crabill, B. (Montague Water Conservation District). Personal communication. 4/18/2007.

Deas, M.L., A. Abbott, A. Bale. 2003. Shasta River Flow and Temperature Modeling Project. Prepared for Watercourse Engineering, Inc. Sponsored by the Shasta Valley Resource Conservation District with funding by CDFG from the CA. Coastal Salmon Restoration Program. Napa, CA.

Deas, M.L., P.B. Moyle, J. Mount, J.R. Lund, C.L. Lowney, S. Tanaka. 2004. Priority Actions for Restoration of the Shasta River – Technical Report. Prepared for The Nature Conservancy. June 11.

Deas, M.L. 2006. Big Springs Creek and Spring Complex – Estimated Quantification Technical Memorandum. Prepared for the North Coast Regional Water Quality Control Board. Davis, CA.

Deas, M.L. Watercourse Engingeering, Inc. Personal communication. 1/2008.

DWP-DWR (Dept. of Public Works, Division of Water Rights, State of California). 1925. Water Supply and Use of Water from Shasta River and Tributaries, Siskiyou County, California. In Shasta River Adjudication Proceedings.

FAO (Food and Agriculture Organization of the United Nations). 1989. Irrigation Water Management: Irrigation scheduling. Available online: http://www.fao.org/docrep/T7202E/t7202e08.htm. Accessed 1/2007.

Geisler, L. 2005. Shasta River Flow, Temperature, and Dissolved Oxygen Model Calibration Technical Report. Prepared for ICE, UC Davis and NCRWQCB. UC Davis and Watercourse Engineering, Inc. Available online: http://www.waterboards.ca.gov/northcoast/programs/tmdl/shasta/060707/20AppendixDShastaRiverFlowTemperatureandDissolvedOxygenModelCalibrationTechnicalReport.pdf. Accessed 5/2007.

Hauser, G.E., G.A. Schohl. 2002. River Modeling System v4 – User Guide and Technical Reference; Report No. WR28-1-590-164; TVA River System Operations and Environment. Norris, Tennessee.

Jeffres, C. (U.C. Davis Watershed Sciences Center). Personal communication. 4/22/2008.

Jeffres, C., E. Buckland, J. Kiernan, A. King, A. Nichols, S. Null, J. Mount, P. Moyle, M. Deas,. 2008. Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California – Water Year 2007. Prepared for the California Nature Conservancy by U.C Davis Center for Watershed Sciences and Watercourse Engineering, Inc.

Lowney, C.L. 2000. River Temperature Dynamics in the Sacramento River: Measuring and Monitoring. Ph.D. dissertation. U.C. Davis Dept. of Civil and Environmental Engineering.

Martin, J.L., S.C. McCutcheon. 1999. Hydrodynamics and transport for water quality modeling. Lewis Publishers. New York, NY.

Moyle, P.B. 2002. Inland Fishes of California. University of California Press. Berkeley and Los Angeles, California.

NCRWQCB (CA. North Coast Regional Water Quality Control Board). 2004. *Shasta River Water Quality Conditions: 2002 & 2003* – Draft for Public Review. May.

NCRWQCB (CA. North Coast Regional Water Quality Control Board). 2006. Staff Report for the Action Plan for the Shasta River Watershed Temperature and Dissolved Oxygen Total Maximum Daily Loads. Santa Rosa, CA. Available online (http://www.swrcb.ca.gov/rwqcb1/programs/tmdl/shasta/060707/staff-report-complete.pdf). Accessed 6/2007.

NRC (National Resources Council). 2004. Endangered and Threatened Fishes in the Klamath River Basin: Causes of Decline and Strategies for Recovery. Committee on Endangered and Threatened Fishes in the Klamath River Basin, National Research Council. The National Academy Press.

Sear, D.A., P.D. Armitage, F.H. Dawson. 1999. Groundwater dominated rivers. Hydrological Processes. 13, 255-276.

TNC (The Nature Conservancy). 2005. A Strategy for Enhancing Flow and Protecting Anadromous Fish Habitat in the Shasta River. Prepared by Kat Ridolfi.

TNC (The Nature Conservancy). 2006. Nelson Ranch 9-inch resolution, truecolor, aerial imagery. Obtained from C.Babcock and A.Fowler.

Scott, J. (Shasta Valley Watermaster). Personal Communication. 4/13/2007.

Vignola, E., M.L. Deas. 2005. Lake Shastina Limnology. Watercourse Engineering, Inc. Prepared for the UC Davis Information Center for the Environment and the North Coast Regional Water Quality Control Board. Davis, CA.

Webb, D. (Shasta Valley Resource Conservation District). Personal Communication. 7/2006.

Welsh, H.H., G.R. Hodgson, B.C. Harvey, M.E. Roche. 2001. Distribution of Juvenile Coho Salmon in Relation to Water Temperatures in Tributaries of the Mattole River, California. North American Journal of Fisheries Management. 21:464-470.

Chapter 6: Systems Analysis for Environmental Water Management

This chapter examines the potential to improve fish habitat conditions by better managing environmental water quantity and quality. Optimization modeling is used to maximize out-migrating smolts from a natal stream, based on coho salmon in California's Shasta River. Restoration activities altering flow and water temperature conditions are the decisions variables of the model, and include increasing instream flow, relocating the Grenada Irrigation District (GID) diversion, increasing riparian shading, restoring the Big Springs Complex, and removing Dwinnell Dam. One fish species is modeled here, although this approach could be applied to various environmental objectives.

Modeling increases understanding of the interaction of physical habitat, limiting factors, and fish population dynamics for management purposes, such as whether particular weeks or life stages produce bottlenecks limiting out-migrating smolts, and which restoration options provide the most habitat improvement for a given restoration budget. The modeling undertaken here provides one example of an approach to quantify instream habitat, evaluate the effects of proposed restoration actions, and manage limited environmental water and budget allocations efficiently and creatively.

This chapter begins with a literature review and overview of the Shasta River, followed by a description of the model, including formulation, data sources, and discussion of decision variables and economic costs. Results are discussed and the limitations of this approach are outlined. A section follows on model improvements and additional applications for which this type of work would be useful. Major conclusions are discussed at the end.

Systems Modeling and Literature Review

Optimization is an approach to systems analysis that explicitly seeks the 'best' solution to a problem within constraints. It helps a decision maker identify a better course of action than might otherwise have been found for complex problems when flexibility exists in the system (Labadie, 2004). An objective function expresses the goal of the model, which is maximized or minimized to arrive at an optimal solution. Constraints define the feasible region. The objective function and constraints are mathematical functions of decision variables and parameters. Decision variables are changeable values which are decided by the model, and parameters are given (Hillier and Lieberman, 1967; Cohon, 1978). Linear optimization is the most common form, although ecosystem dynamics are often non-linear.

Systems analysis for water resources has historically focused primarily on simulation and optimization of human water uses including urban and agricultural water supply reliability, flood control, hydropower generation, and to a lesser extent, recreation uses (Sale et al., 1982; Cardwell, 1996). When environmental objectives are included in systems analysis, they are typically modeled as constraints to remove them from economic valuation and decision-making (Draper et al., 2003). However, modeling studies that include competing water uses, such as environmental objectives with traditional human-based objectives, are becoming more common and increasingly needed as systems are operated more tightly for urban and agricultural efficiency, hydropower, environmental sustainability, fisheries production, and water quality (Labadie, 2004).

Including environmental quality objectives in systems analysis and mathematical modeling began in the 1960s, with optimization of dissolved oxygen targets for the Willamette River from a waste treatment plant (Liebman and Lynn, 1966). In the 1980s, instream flow needs were incorporated into a reservoir optimization model (Sale et al., 1982), and tradeoffs between instream fish flows and municipal water supplies were quantified using a computer simulation model (Palmer and Snyder, 1985). Cardwell (1996) used multi-objective optimization to improve water reliability and fish habitat in a simple reservoir-stream system. Higgins and Brock (1999) incorporated minimum flow and dissolved oxygen targets into the historic Tennessee Valley Authority (TVA) operating priorities of navigation, flood control, and hydropower production. Multi-objective optimization of water supply and instream flow objectives was used to demonstrate a framework for evaluating the tradeoff between instream and human water needs (Homa et al., 2005). Watanabe et al. (2006) used simulation and optimization modeling to determine efficient allocation of management activities to protect salmon populations and decrease water temperatures. Optimization has also been used to assess habitat value and assist decision making in non-water resource systems, such as maximizing the geographic coverage of protected migratory bird stopovers (Williams, ReVelle, Bain, 2003).

Considering environmental quality in river systems typically involves instream flow requirements. Simulation modeling is the most common method of measuring the effects of instream flows, although little insight is gained for making decisions among competing water uses. The need for better instream flow methodology has been well documented (Richter et al., 1997) and numerous techniques for determining instream flow requirements exist (Jowett, 1997). Optimization modeling has been used for theoretical examples, but has yet to be widely implemented due in part to skepticism, mathematical complexity, software limitations, and period-of-record solutions rather than updated rule curves for reservoirs (Labadie, 2004). However, optimization provides a worthwhile method to weigh decisions, eliminate poor alternatives, and highlight promising solutions (Null and Lund, 2006).

Coho and the Shasta River

The Shasta River is located in Siskiyou County, California, and is the last major tributary to the Klamath River before Iron Gate Dam, the first dam on the Klamath River. The Shasta River currently has a mean annual flow of 138 taf, with substantial springwater flows, and precipitation inflows. The Shasta River has one major dam, Dwinnell Dam at river mile 40.62, and numerous small diversion dams.

Historically, the Shasta River was a highly productive salmon stream with fall run Chinook, spring run Chinook, coho, steelhead trout, as well as non-salmonids such as Klamath River and Pacific lamprey, speckled dace, smallscale sucker, and marbled sculpin. Spring run Chinook were extirpated with construction of Dwinnell Dam. Fall run Chinook, coho, and steelhead remain in the Shasta River with depleted numbers and coho have been listed as a federally threatened species (Moyle, 2002).

Today fish productivity in the Shasta River is limited by low flow conditions and increased water temperature. Low flow conditions are due to surface water diversions, groundwater pumping, and construction of Dwinnell Dam. Increased water temperature is primarily from low flows, loss of riparian vegetation, tailwater return flow, and diversion of cooler springwater inflows (DWR, 2001; NRC, 2004). Additional habitat problems exist, such as gravel recruitment, access to spawning and rearing habitat, barriers to migration, low dissolved oxygen, and turbidity.

Methods

Model Description

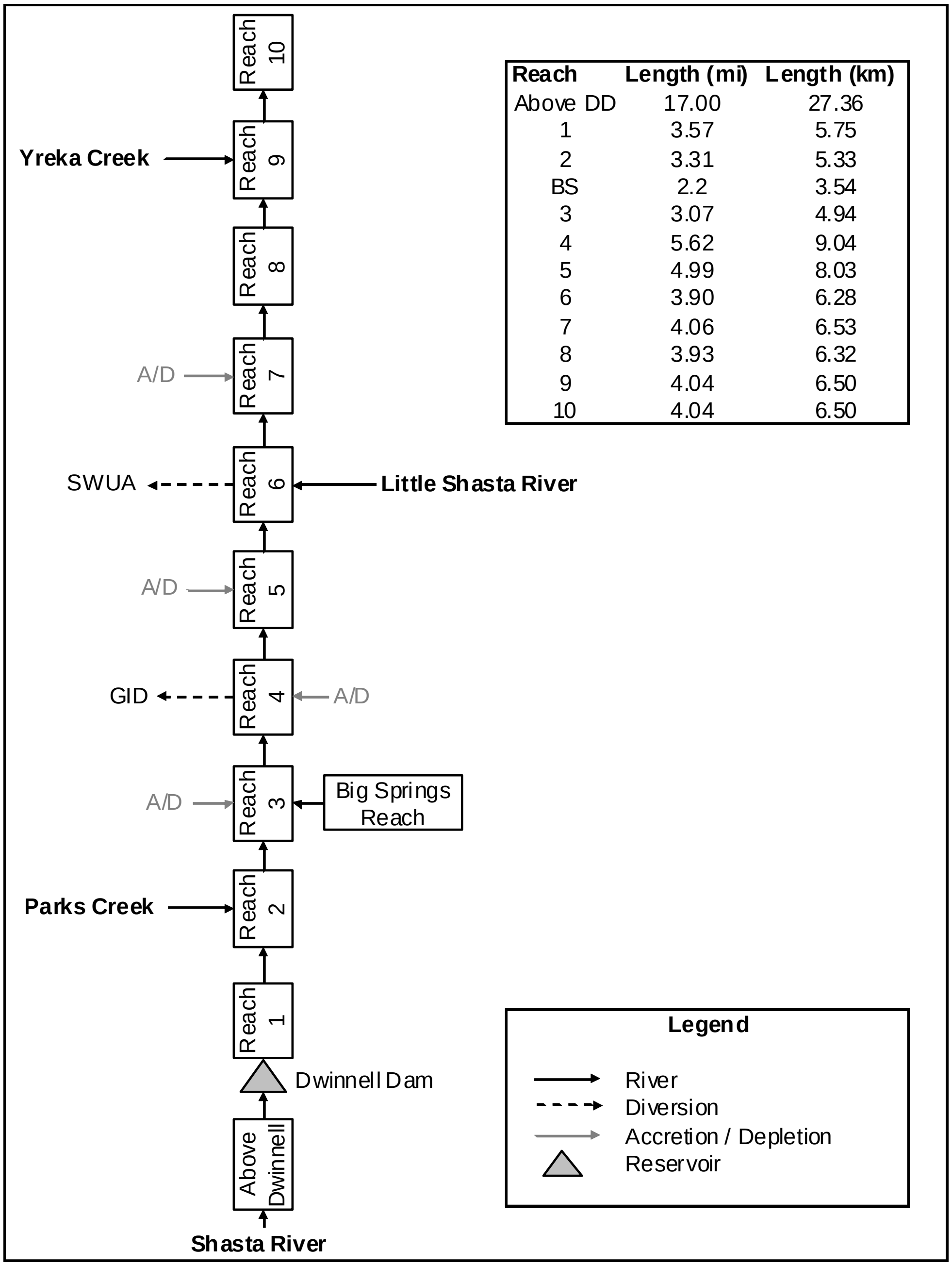
The fish habitat optimization model is a one-dimensional model of network flow along the Shasta River, where flow and water temperature change longitudinally by reach but are assumed to be well-mixed laterally and vertically. Escapement of one fish species, based on the life history of coho salmon, is maximized. The model is constrained by conservation of mass and heat (energy), habitat capacity as a function of instream flow and water temperature, fish demography, restoration budget, and upper and lower bounds for instream flow and fish by reach and age class.

Three fish life stages are modeled: alevin, juvenile rearing, and out-migrating smolts, and each have distinct timing and optimal flow and temperature requirements (discussed further in the input data section of this chapter). The model operates on a weekly time step, with weekly averaged flow and water temperature data from 2001 RMSv4 simulations of the Shasta River (Null, 2008). The model runs in Microsoft Excel using Lindo Systems What's Best commercial solver (Lindo Systems Inc., 2005). Other habitat quality considerations, including substrate, barriers to migration, instream cover, other water quality conditions, Klamath River conditions, and ocean conditions are ignored.

The Southern Oregon/Northern California Coast ESU of coho salmon was listed as federally threatened by NMFS in 1997. Under the Endangered Species Act, the federal government is now legally required to provide and protect critical habitat for this stock of coho. Critical habitat is the area (including water, substrate, and adjacent riparian zone) essential to the conservation of the species which may require special management or protection (US EPA, 2008).

Often environmental goods, such as fish stocks, are valued economically in mathematical modeling. Although methods of quantifying economic values of environmental goods exist (Loomis, 2000), the model described here was formulated to avoid economic valuation of fish production or fish habitat. In this case, the government is required to protect coho and their habitat. Thus, maximizing out-migrating smolt is the objective (rather than minimizing costs), while the costs of proposed restoration activities form the budget constraint and are more readily valued. Additionally, the model described here is nonlinear because some heat budget and habitat capacity constraints are nonlinear.

The river is divided into twelve reaches, ten in the mainstem Shasta River, one above Dwinnell Dam, and one in Big Springs Creek (Figure 112). For each reach initial instream flow and water temperature conditions, flow and temperature boundary conditions at tributaries, diversions (including accretions/depletions), and atmospheric heating determine water and heat budgets (Figure 113). Atmospheric heating rates are applied occurs during summer, although the rate of heating depends on the extent of riparian shading present (volume of water in the system, as well as headwater and tributary inflow and temperature are ignored here). Flow and water temperature at each reach and time step then determine the habitat capacity for alevin, juveniles, and out-migrating smolts.



Reach	Length (mi)	Length (km)
Above DD	17.00	27.36
1	3.57	5.75
2	3.31	5.33
BS	2.2	3.54
3	3.07	4.94
4	5.62	9.04
5	4.99	8.03
6	3.90	6.28
7	4.06	6.53
8	3.93	6.32
9	4.04	6.50
10	4.04	6.50

Figure 112. Shasta River model schematic with reach lengths

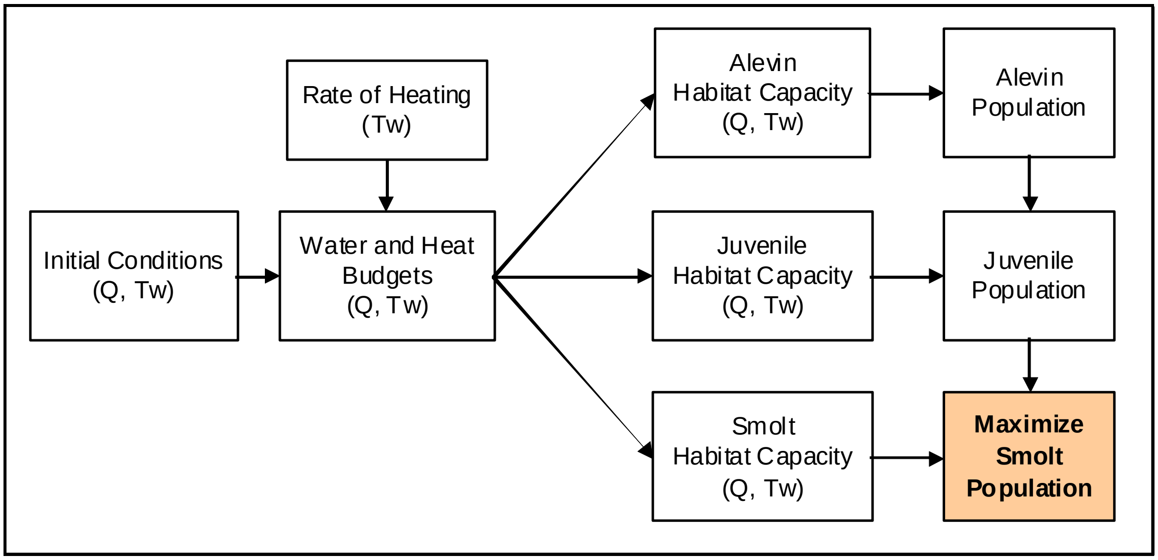


Figure 113. Fish production model flow chart

Proposed restoration alternatives for the Shasta River are decision variables of the model, and include adding flow to any reach, moving the GID diversion, increasing riparian shading, restoring the Big Springs Complex, and removing Dwinnell Dam. Each restoration alternative alters instream flow and temperature conditions. All restoration activities have been simulated in the previous chapter to understand how flow and water temperature change with alternatives, and to generate needed input data (Null, 2008).

Additional proposed methods to improve Shasta River instream habitat conditions that are not included in this study include: tailwater management, conjunctive use strategies, and water transfers from the Klamath River. Tailwater returns to the Shasta River over its length, although the cumulative effects on instream water temperature have not been quantified. Likewise, groundwater in the Shasta Valley has not been thoroughly studied, and in general is poorly understood. Water transfers from the Klamath River are not included because they are likely prohibitively expensive. (Assuming 630 ft [192 m] between the Klamath River below Iron Gate and Dwinnell Dam and energy efficiency of 70%, pumping would use 921.7 kWh/af. If energy costs $0.07/kWh, then pumping alone costs $64.52/af.)

Formulation

Computer optimization is used to maximize the number of smolts out-migrating from the Shasta River.

$$MaxF = \sum_{w}\sum_{r} F_{a=3,w,r} \qquad \text{Maximize smolts} \qquad \textbf{(53)}$$

where $F_{a,w,r}$ is fish, a is fish age class (age class three are smolts), w is week, and r is reach.

subject to:

$$Q_{w,r+1} = Q_{w,r}(RD) + b_{w,r}, \forall w,r \qquad \text{Conservation of mass} \qquad \textbf{(54)}$$

$$T_{w,r+1} = \frac{Q_{w,r}(RD) * T_{w,r}(RD)}{Q_{w,r}(RD)} + \Delta T_{w,r}, \forall w,r$$ Conservation of heat **(55)**

$$HC_{a=1,w,r} = \left(C * \frac{1}{(\sigma\sqrt{2\pi})}\right) * \left(\frac{e^{-(Q_{w,r}-\mu)^2}}{2\sigma^2}\right) * \left(\frac{k^n}{k^n + T_{w,r}^n}\right)$$ Alevin Habitat Capacity **(56)**

$$HC_{a,w,r} = \frac{\left(\frac{k^n}{(k^n + T_{w,r}^n)}\right)}{(1 + e^{d-g*Q_{w,r}})}$$ Juvenile and Smolt Habitat Capacity **(57)**

$$F_{a,w,r} \le HC_{a,w,r} * (\alpha_{a=1,w,r} * x_{w,r} * d_{a=1,w,r}) * \beta_{a=1,w,r}, \forall a,w,r$$ Alevin Fish Production **(58)**

$$F_{a,w,r} \le HC_{a,w,r} * (\alpha_{a,w,r} * x_{w,r}), \forall a,w,r$$ Juvenile and Smolt Fish Production **(59)**

$$\sum_r F_{a,w,r} \le \sum_r F_{a,w-1,r} + (\sum_r F_{a=1,w-4,r} * \theta_{a-1,w,r}), \forall a,w$$ Juvenile Demography **(60)**

$$\sum_w F_{a,w,r} \le \sum_w F_{a,w,r-1} + (F_{a-1,w-1,r} * \theta_{a-1,w,r}), \forall a,w,r$$ Smolt Demography **(61)**

$$F_{a,w,r} \ge F_{a,w,r-1}, \forall a,w,r$$ Smolt Downstream Access **(62)**

$$l_{w,r} \le Q_{w,r} \le u_{w,r}, \forall w,r$$ Flow capacity bounds **(63)**

$$l_{a,w,r} \le F_{a,w,r} \le u_{a,w,r}, \forall a,w,r$$ Fish capacity bounds **(64)**

$$B \ge \sum_w \sum_r c_{w,r} * RD$$ Restoration budget **(65)**

where $Q_{w,r}$(RD) is flow from a given restoration decision; $b_{w,r}$ is additional inflow or outflow; $T_{w,r}$(RD) is water temperature from a given restoration decision; $HC_{a,w,r}$ is habitat capacity as a function of flow and water temperature; C is 176 (a constant), σ (variance) is 70, μ (mean) is 180, and k, n, d, and g are parameters with values of 15, 20, respectively for alevin; 16, 20, 4, 0.1, respectively for juvenile; and 17, 22, 5.3, 0.13, respectively for smolt (this is described in further detail in the habitat logistic surface section below); $\alpha_{a=1,w,r}$ is maximum number of fish per redd (Table 21); $x_{w,r}$ is the length of the reach; $d_{a=1,w,r}$ is the maximum number of redds per river mile (Table 20); $\beta_{a=1,w,r}$ is a parameter shaping the timing of emergence into a bell curve (Figure 115); $\theta_{a-1,w,r}$ is survival of one life stage to the next (1- mortality); $u_{a,w,r}$ is an upper bound; $l_{a,w,r}$ is a lower bound; B is total restoration budget; $c_{w,r}$ is restoration cost; and RD is a restoration decision.

Each restoration decision affects flow and/or water temperature (Table 19), which in turn changes the number of surviving smolts. The data and assumptions used for each restoration alternative and figures illustrating changes to instream flow and thermal conditions are discussed further in the input data section of this chapter.

Table 19. Habitat model decision variables and assumptions

Decision Variable	Policy Activity	Modeled Effect	Cost
Additional Flow	Reduce diversions, dam releases, water markets	Increase Q in any reach	$900/cfs-week (TNC, 2005 estimates $36-66/af)
Move GID	Move GID diversion to Hwy A-12	Increase Q for 6.5 mi. Locally reduce Tw	Assumed at $1 million
Riparian Shading	Actively replant riparian vegetation	Decrease Tw (reduce atmospheric heating)	$6758/mile for conservation planting (Quinn et al., 2001)
Restore Big Spring Creek	Buy Big Springs property / water rights	Increase Q, decrease Tw (preserve cold spring Tw, greater thermal mass)	$15 million (purchase Busk property) (TNC)
Remove Dwinnell Dam	Remove Dwinnell Dam	Increase Q, decrease Tw (greater thermal mass, cooler initial Tw)	Assumed at $15 million (not including water replacement)

The constraints needed in each life stage to define upper and lower bounds of population cohorts and move fish from one age class to the next change slightly by age class, and are discussed below. All age classes of fish are modeled as adjustable values so the maximum number survive under multiple constraints, and the model can place fish in the best reaches to ensure optimal production (the model is optimistic in these ways). Additionally, values of fish are continuous rather than integers to improve model run time.

Alevin

For the alevin age class, habitat capacity provides the upper bound, and non-negativity provides the lower bound. There is no demography constraint for this age class because it is the first cohort modeled. Alevin are assumed to remain near redds, and thus cannot move between reaches. Alevin habitat does not exist in reaches 6 – 9 (river mile 4.05 – 19.98), because fish have not been known to spawn there (Table 20). Coho primarily spawn in the canyon reach (RM 0 – 4.3), and near the Big Springs Complex (RM 34 – 40.62), although limiting spawning may occur below Big Springs near Nelson Ranch (RM 27.5 – 32) (Jeffres et al., 2008). Redds per mile can be highly variable depending on characteristics of each river system. The numbers used here were estimated with the expert opinion of a fish biologist studying the Shasta River (Jeffres, pers.comm., 2008).

Table 20. Length, location, and maximum number of redds by reach

Reach	Length (mi)	Location (RM)	Redds/Mile
Above Dwinnell Dam	17.00	40.63 – 57.63	50
Reach 1	3.57	37.05 – 40.62	50
Reach 2	3.31	33.73 – 37.04	50
Big Springs Reach	2.2	BS source – Reach 3	50
Reach 3	3.07	30.65 – 33.72	20
Reach 4	5.62	25.02 – 30.64	1
Reach 5	4.99	20.02 – 25.01	1
Reach 6	3.90	16.11 – 20.01	0
Reach 7	4.06	12.04 – 16.10	0
Reach 8	3.93	8.10 – 12.03	0
Reach 9	4.04	4.05 – 8.09	0
Reach 10	4.04	0 – 4.04	50

The number of alevin per redd is estimated from Moyle (2002), who states females lay 1400-3000 eggs per redd, and mortality is 10% after hatching under optimal conditions (Table 21). Further considering suboptimal occurs in the model as additional fish die from inadequate flow and water temperature conditions.

Table 21. Values for α, Maximum Number of Individuals; θ, mortality of fish until smolt life stage, and timing by life stage

Age Class	Maximum Fish, α	Mortality, θ (%)	Timing (wks)	Timing (dates)
Alevin	2000 (per redd)	8	1 – 22	1/1 – 6/3
Juvenile	400 (per river mile)	10	5 - 5	1/29 – 2/4
Smolt	90 (per river mile)	0	6 - 22	2/26 – 7/1

Mortality estimates for coho populations from predation, competition, and food abundance (density dependent) were limited. Nickelson (1992) estimates density independent mortality at 68% for alevin and 70% for juveniles; however these estimates already incorporate habitat quality. Sensitivity analysis on mortality rates was completed. With mortality of 8% and 10% for alevin and juveniles, respectively, 99 fish survive under current conditions (Table 21). No accurate escapement numbers exist for the Shasta River because counting traps are removed with high flows when coho out-migrate (CDWR, 2003b). 99 fish is probably a low estimate given further mortality from Klamath River and ocean conditions. However given the uncertainty of input data, for this study results should be interpreted by relative survival between restoration options, rather than absolute numbers of fish survival. The model is sensitive to the value of the mortality parameter, it could be used to further calibrate the model given accurate escapement data.

Alevin are assumed to be in the Shasta River from January through May (weeks 1 – 22) (Figure 114), although their numbers are limited by β, a parameter shaping their distribution through time, so most alevin emerge in March (Figure 115) (CDFG, 2002;

NCRWCQB, 2006). Four weeks after emergence, fish move from the alevin stage to the juvenile rearing stage. For this model emergence is not temperature dependent, although in reality emergence is highly correlated with water temperature.

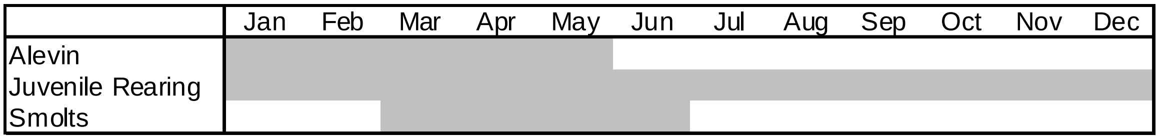


Figure 114. Modeled age class timing (DFG, 2002; SSRT, 2003; NCRWQCB, 2006)

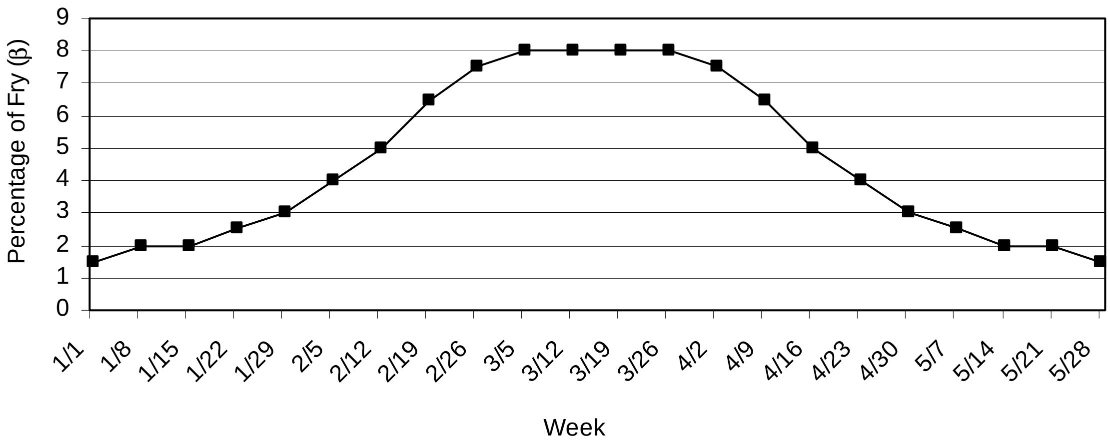


Figure 115. Timing of alevin emergence, β

Juvenile Rearing

In the juvenile stage, fish are assumed to move freely between reaches. In addition to upper and lower bounds, fish demography constraints limit the maximum number of fish in each life stage; the total number of fish in any life stage cannot exceed those in the previous life stage. Because coho rear for a full year (February to February), the total number of juvenile fish for all reaches cannot increase through time, with the exception of fish entering from the alevin stage.

The number of juvenile fish must be less than habitat capacity times the maximum number of fish per river mile (α_a) (Table 21) times reach length for all weeks and reaches. Nickelson (1992) estimates potential population of juvenile coho is between 54 - 3444 fish/mi in Oregon coastal streams. The stream with 3444 fish/mi (Benson Creek) is an anomaly, the stream with next highest population potential is 1000 fish/mile. Ignoring Benson Creek, average population potential for juvenile coho is 489 fish/mi. For this study, 400 fish/mi was used as a conservative estimate. Habitat capacity is multiplied by number of fish per river mile and reach length to preserve the spatial component of the model, since reaches have different lengths. The demography constraint maintains that juveniles in all reaches must be fewer than juveniles in the previous week plus incoming alevin multiplied by mortality.

Smolts

Smolts out-migrate from late February through June (Figure 114) (CDFG, 2002; NCRWQCB, 2006; SSRT, 2003). Out-migration can be completed within a single week, or smolts can hold in any reach through the out-migration season to wait for more favorable conditions, but cannot return upstream. Additionally, in the smolt stage fish must swim downstream toward the Klamath River through all river reaches, without skipping reaches, requiring adequate habitat capacity in downstream reaches. There can be no more smolts than there were juveniles in the final juvenile rearing time period. In reality, increased flows provide cues for smolts to out-migrate (Moyle, 2002), although modeling here is based on average timing of coho out-migration in the Shasta River (SSRT, 2003).

The habitat and non-negativity constraints are unchanged from the juvenile age class. Smolt demography ensures that total smolts for all weeks in a reach cannot exceed the previous reach plus incoming juvenile fish for a given reach times mortality. The downstream access maintains that smolts for all weeks and reaches must be less than the previous reach. The maximum number of fish per reach was estimated using the method discussed for the juvenile age class (Nickelson, 1992).

Habitat Logistic Surfaces

Ideal water temperatures and velocities for coho salmon for each modeled life stage are listed in Table 22. These values and the expert opinion of a fish biologist who works on the Shasta River (Jeffres, pers.comm., 2008) were used to build habitat logistic surfaces linking instream flow and temperature conditions with coho survivorship (Figure 116). The habitat capacity surface for alevin uses equation 56, and the habitat capacity surface for juveniles and smolts use equation 57.

Table 22. Ideal water temperature and velocity for coho by life stage (Moyle, 2002; CDFG, 2002; CBSED, 2005)

Age Class	Ideal Temperature (ºC)	Ideal Velocity (ft/s)
Alevin	4 – 13	< 0.5
Juvenile	12 – 14	< 1.0
Smolt	8 - 16	Timing influenced by pulse flows

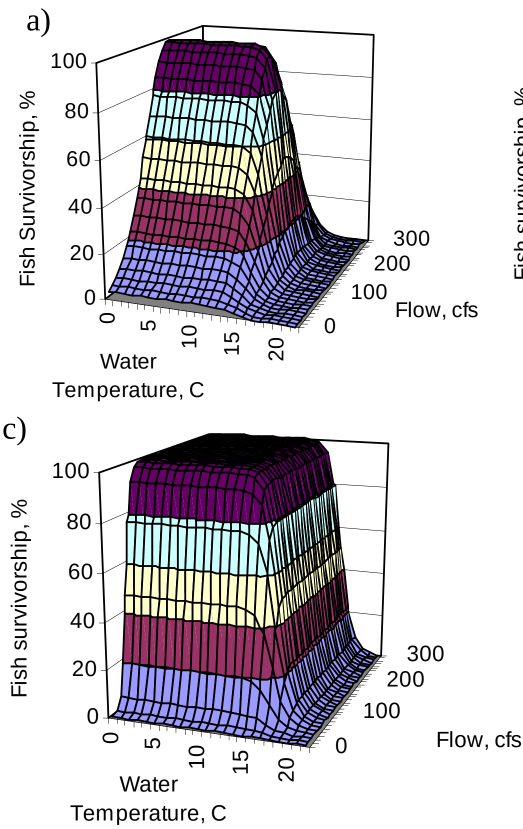


Figure 116. Flow and water temperature habitat logistic surfaces for coho a) alevin, b) juveniles, and c) smolts

Although velocity is a more common metric of fish health than flow (CDSED, 2005), it was assumed that the ideal velocity values in Table 22 exist in all reaches for this study. It has been documented that during summer and early fall, a variety of velocities are available in the Shasta River because abundant macrophyte growth provides mid-channel low velocity refuge for fish (Jeffres et al., 2008). The Shasta River is unique because typical coho habitat, such as large woody debris is uncommon. Aquatic macrophytes may provide substantial seasonal habitat in some reaches.

Instream habitat for the alevin age class varies with a bell-shaped curve with flow conditions, so 100% of alevin survive at 180 cfs and survivorship drops with more or less flow (Figure 116). Low flows can expose and desiccate redds, while high flows can wash away hatchlings or mobilize redd gravels (CDFG, 2002). In the model, water temperature affects habitat through a logistic relationship. From 0 - 11ºC, 100% of alevin survive, and mortality quickly rises with increased temperatures. At 15ºC, 50% of alevin survive, and by 18ºC, only 3% of alevin survive. Equation 56 was used to create the alevin habitat logistic surface in Figure 116.

The juvenile and smolt habitat logistic surfaces are both logistic surfaces and are similar (Figure 116). 100% of fish survive when flow reaches 80 cfs or more for juveniles and smolts. Survivorship drops at slightly lower temperatures for juvenile coho than smolts. For juvenile fish, survivorship is 98% at 13ºC, 50 % at 16ºC, and 3% at 19ºC. For smolts, survivorship is 94% at 15ºC, 50% at 17ºC, and 3% at 20ºC. For both age classes, water temperature would be more accurately modeled with a trapezoidal shaped curve because survivorship could near 100% for a range of temperatures (Table 22). Also, survivorship should decrease with temperatures below 8ºC and 12ºC for juveniles and smolts, respectively. However, it is assumed that much of the Shasta River is partially spring fed and thus maintains higher winter water temperatures, so a logistic surface with no decrease in survivorship with low water temperature was used.

Flow and Water Temperature Input Data (Year 2001)

As stated above, the Shasta River is a spring-dominated river above approximately river mile 20, but is runoff-dominated from local precipitation, runoff, and spring snowmelt during the winter and spring. In general, groundwater-dominated river systems have a more stable flow and thermal regime than runoff dominated rivers (Caissie, 2006). There is little loss in data quality when flow is averaged into weekly values because of the stability of the springwater contributions, although peaks from winter storm-related pulses in flow are lost (Figure 117). Data for all modeled reaches (Shasta River, Big Springs Creek) and boundary flows (Parks Creek, Little Shasta River, and Yreka Creek) were averaged from daily flow data used in the current conditions RMS4 simulation model. Raw flow data for the simulation model was from NCRWQCB, USGS, and water balance calculations (Null, 2008). Big Springs flow was measured by UC Davis from 3/26 – 6/5/2008. Data from this time period was aggregated to weekly values for initial flow, with the remaining time series estimated from UC Davis and NCRWQCB data.

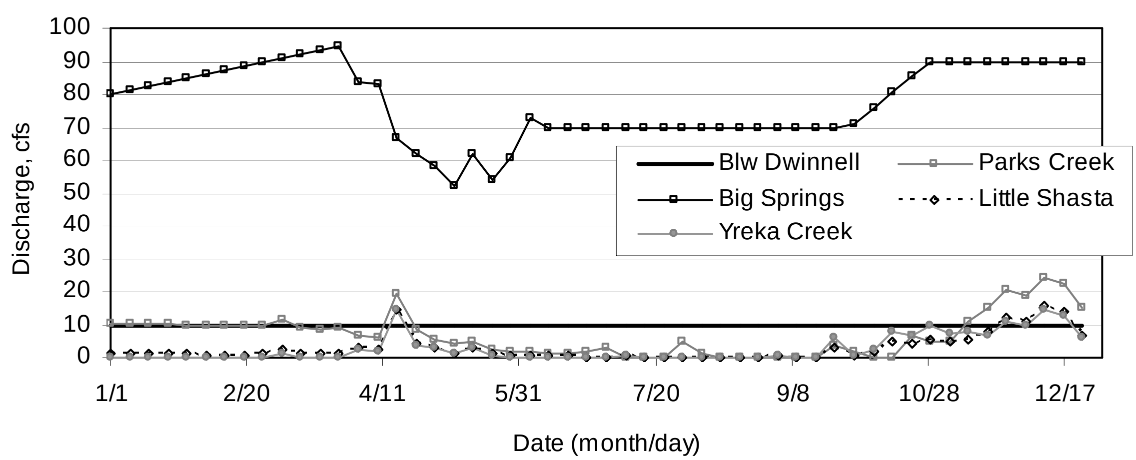


Figure 117. Hydrology initial conditions of modeled reaches and boundary flows

The GID and SWUA diversions are modeled as withdrawals (Figure 118). Diversions are estimated on a seasonal scale (average volume of withdrawals), thus aggregation to weekly values does not affect data quality (in reality irrigators divert on a daily timeframe, but detailed data are not available). Additionally, accretions and depletions representing small diversions, tailwater return, subsurface flow, and local precipitation are added to reaches three, four, five, and seven so the water balance closes

(Figure 118). Accretion and depletion timeseries were obtained from current conditions simulation results for the midpoint of all ten mainstem reaches in the optimization model. Where flow discrepancies existed between reaches, the difference was added to the optimization model as accretions and depletions.

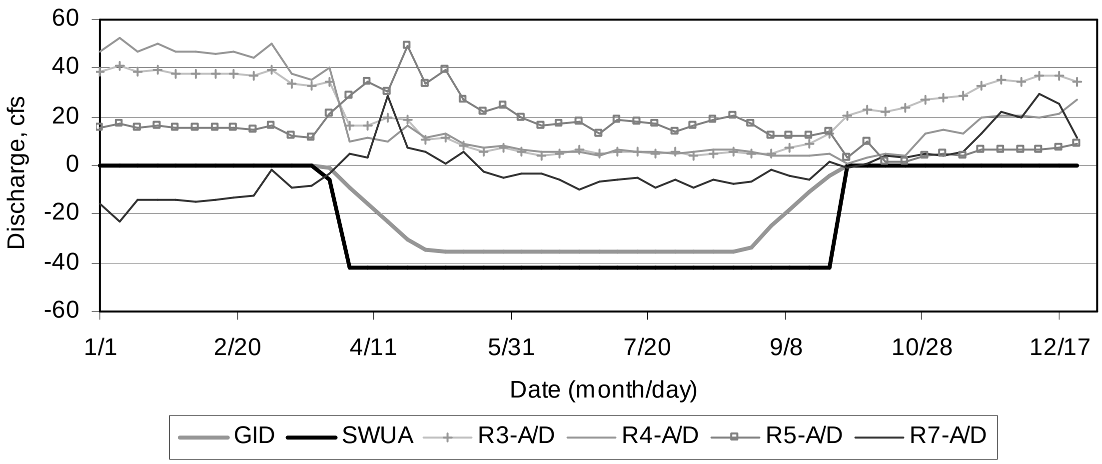


Figure 118. Diversions and accretion / depletion input data

Averaging water temperature into weekly values removes all diurnal fluctuations (Figure 119), which are instrumental in the health of cold-water fish species. The effects of daily high temperatures on fish are offset by the length and extent of nightly low temperatures, as well as other habitat criteria such as food abundance (NRC, 2004). However, weekly averaged temperatures are a commonly used metric of fish health (Welsh et al., 2001), and are used here. NCRWQCB 2001 data are used when available, and equilibrium temperature theory was used to estimate temperature data for the remainder of the year (Martin and McCutcheon, 1999; NCRWQCB, 2001; Null, 2008). Measured 2008 water temperature data is used for Big Springs when available (3/26 – 6/25/2008), and temperatures are estimated for the rest of the year.

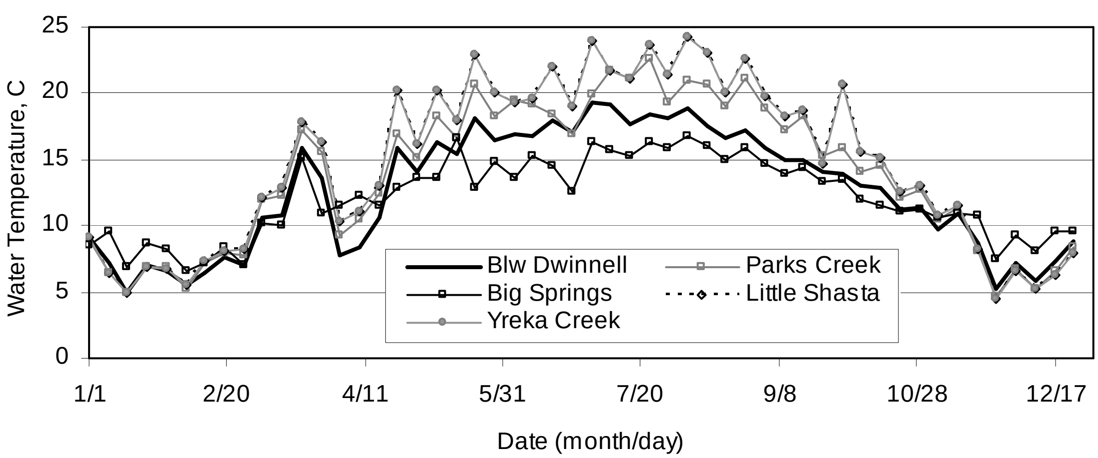


Figure 119. Water temperature input data for initial conditions and boundary flows

Decision Variables and Economic Costs

Water and heat balances are simulated within the optimization model using a mass balance approach to calculate flow and water temperature for all reaches (Figure 120, Figure 121, equations 54, 55). Restoration options for the Shasta River then change instream flow and water temperature conditions, affecting fish habitat, and ultimately the number of out-migrating smolts. Restoration alternatives included in the model are adding flow to any mainstem reach, relocating the GID diversion from its current location to Highway A-12, increasing riparian shading in any reaches below Dwinnell Dam, fully restoring Big Springs Creek, and removing Dwinnell Dam.

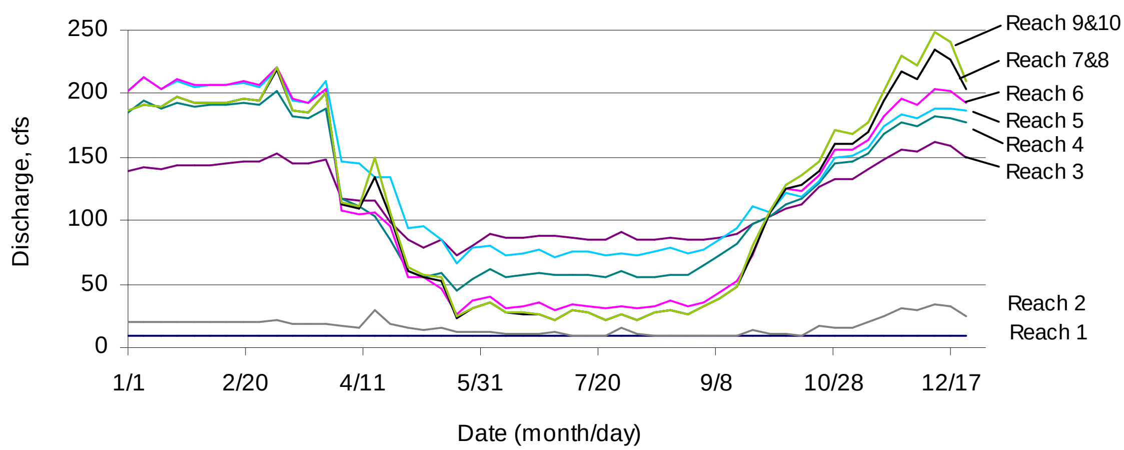


Figure 120. Initial flow in mainstem reaches

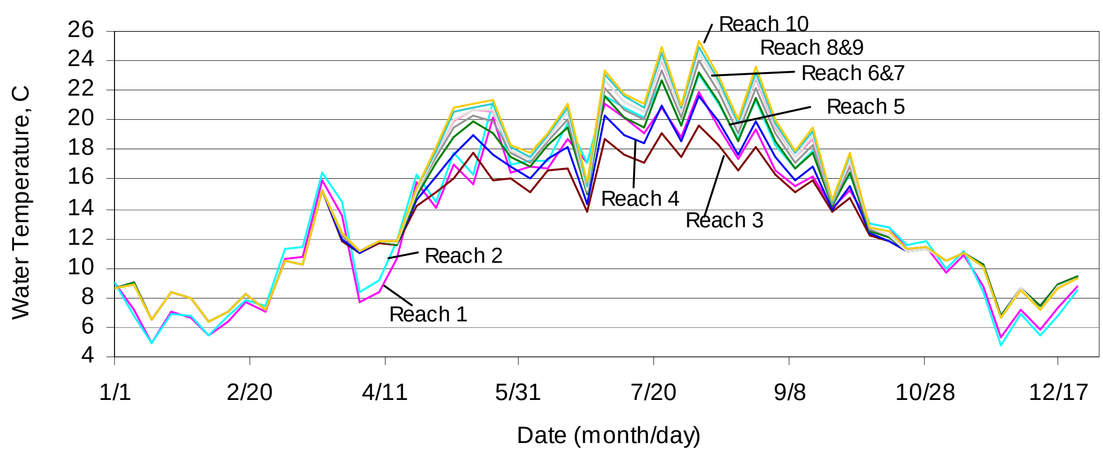


Figure 121. Initial water temperature in mainstem reaches

Additional Flow

Increasing instream flow is commonly recommended to mitigate low flow conditions and reduce water temperature by increasing volume and decreasing travel time, both of which limit atmospheric heating (Deas et al., 2003; CDFG, 2003; TNC, 2005;

NCRWQCB, 2006). For this analysis it is assumed that flow can be added to any reach on the mainstem Shasta River below Dwinnell Dam (reaches 1 – 10), for $900/cfs-week (Table 19). TNC (2005) estimates that temporary leases on water rights cost $36-66/af ($72 – 132/cfs-day, and $500 – 916 cfs-week). For this study $900/cfs-week is used as a conservative estimate for the cost of adding flow to the Shasta River. (Cfs-week is the volume of water required for a constant rate of 1 cfs over a week's time, about 14 af. This is not a common unit, but is necessary here since the model runs on a weekly timestep. Disaggregating weekly data into cfs would connote detail not present in the model.)

Adding flow to the Shasta River could be accomplished by reducing diversions, water markets, or instream flow releases from Dwinnell Dam, although these actions are not distinguished in the model. Additional flow is bounded between zero and the maximum weekly flow from the unimpaired simulation run (Figure 122) (Null, 2008). Extra flow is added at the existing water temperature of the reach, so there are minimal temperature effects from increasing flow. This optimization does not explicitly model physical processes, so changes in travel time and atmospheric heating cannot be directly assessed. Additional flow is modeled as continuous variables, so that any volume between the upper and lower bounds can be added.

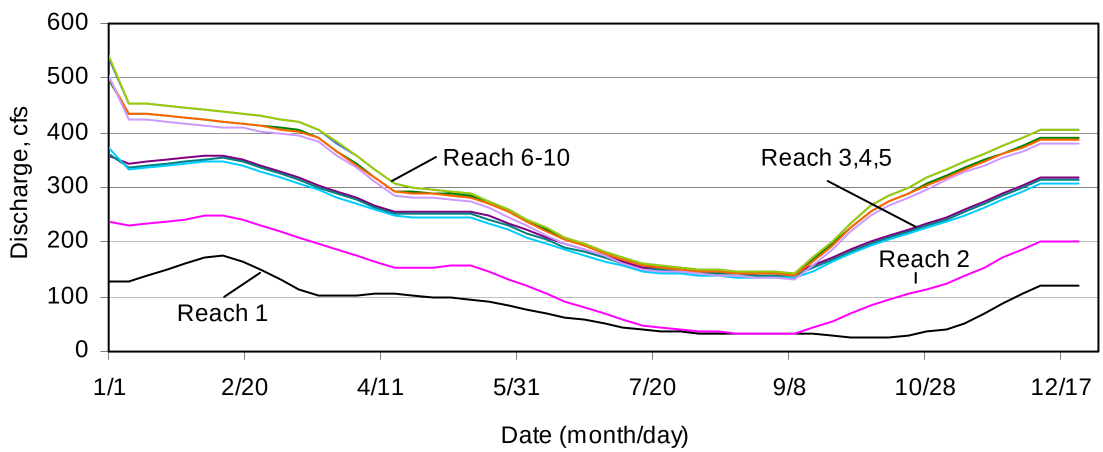


Figure 122. Upper bound for additional flow by reach

Relocation Grenada Irrigation District (GID)

Moving the GID diversion from its current location on the Peter's Ranch in reach 4 (RM 30.58) to Highway A-12 in reach 5 (RM 24.11) maintains flow in the middle-upper reaches of the Shasta River where salmon have been known to spawn (Jeffres et al., 2008). GID has the most junior water right from the Shasta River, and due to pumping costs, its customers pay $52/af of water, the most expensive in the valley (TNC, 2005). TNC has proposed moving GID to extend instream flows while delivering contracted water to customers.

The approximately 35 cfs diversion is simply moved from reach 4 to reach 5 when optimal for fish production in the model. No changes are made to water temperature, although raising flow conditions would likely increase transit time and the air-water interface for 6.5 river miles, and may affect water temperature. This decision is modeled as a binary integer under the assumption the diversion would be moved in its entirety. No cost estimates are available for relocating the GID diversion, so it is assumed to cost $1 million (Table 19).

Increasing Riparian Shading

Increasing riparian shading has long been viewed as a promising method to reduce incoming solar radiation and associated thermal loading on the Shasta River (Deas et al., 2003; NCRWQCB, 2006), and will be particularly effective if paired with other restoration measures that reduce water temperature in the upstream reaches of the Shasta River.

Riparian shading can be added to all reaches below Dwinnell Dam, including Big Springs Creek, to reduce incoming solar radiation. Atmospheric heating associated with solar radiation is based on current conditions and increased riparian shading simulation results, and used as input for the optimization model. Shading reduces heating by reach by the difference in heating rates between the current conditions simulation and riparian vegetation simulation (Figure 123). Heating rates for reach 3 are also used for Big Springs Creek.

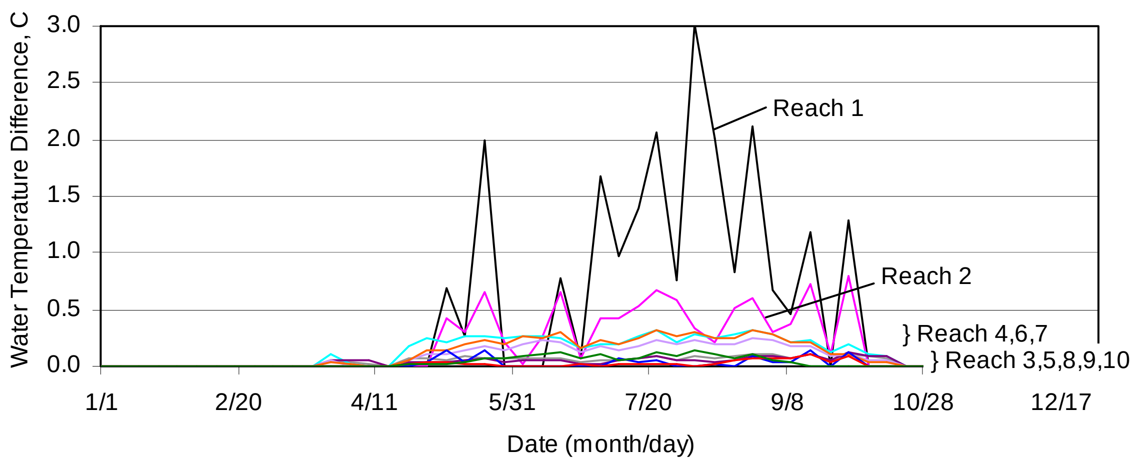


Figure 123. Difference in heating by reach between current conditions and increased riparian shading

Quinn et al. (2001) estimates planting trees costs $6758 (USD) per mile for conservation plants (mangroves, flaxes, and shrubs) in a New Zealand river system (Table 19). Increasing riparian vegetation is not modeled as a binary integer variable, due to long model run times with many binary variables. Rather riparian shading is modeled as a continuous variable between zero and one, with zero representing no additional shading and one representing maximum shading (full riparian restoration). Values between zero and one represent partial shading, such as shading from bulrush and cattail, or widely spaced trees and shrubs.

Restoring Big Springs Creek

Restoring Big Springs Creek increases flow that is cooler than ambient conditions due to the springs influence. This water flows down Big Springs Creek and ultimately enters the Shasta River at reach 3 (Figure 124), where salmon may spawn. For this analysis, restoring the Big Springs Complex is not modeled in detail (i.e., channel restoration, diversion, and return flow management), but assumed that spring waters remain in Big Springs Creek (rather than diverted for adjacent land uses) and is more efficiently conveyed to the Shasta River with reduced rates of heating compared to existing conditions.

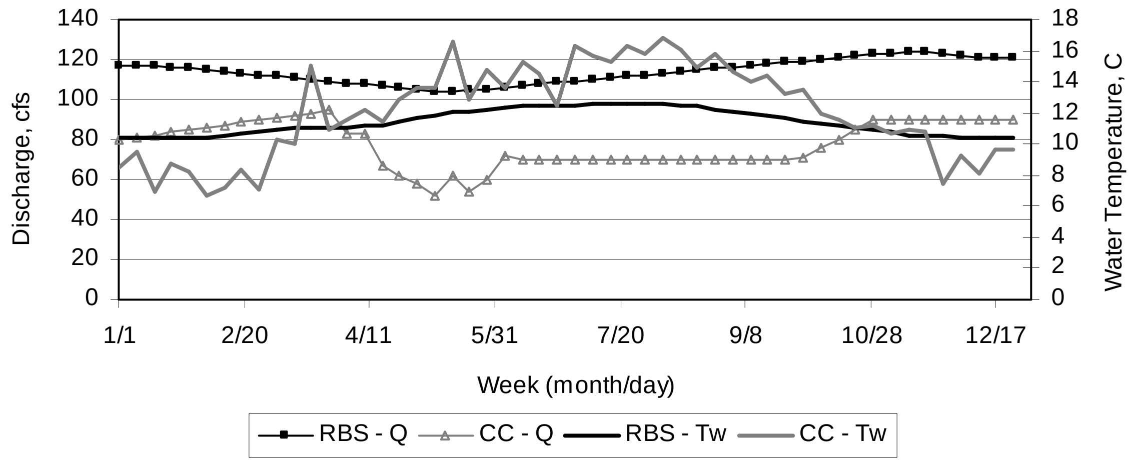


Figure 124. Flow and water temperature at Big Springs with current conditions (CC), and restoring Big Springs (RBS)

Flow and temperature data for restoring Big Springs Creek are mostly from the restored Big Springs simulation results (Null, 2008). Current conditions includes measured flow data from 3/26 – 6/5/2008, measured water temperature data from 3/26/ - 6/25/2008, and uses weekly averaged measured and estimated 2001 flow and temperature from RMS simulations (Null, 2008). TNC (2005) estimates $15 million to buy the Busk property on Big Springs Creek (Table 19). Restoring Big Springs Creek is modeled as a binary integer variable, so it is either restored completely or has current instream conditions.

Removing Dwinnell Dam

Removing Dwinnell Dam has been proposed by the National Research Council (2004) to improve habitat quality below the dam and regain access to reaches above the dam. Flow is increased at the damsite and initial water temperature is cooler as seasonal solar heating no longer occurs in the reservoir. Also, simulation runs providing input data assume upstream tributaries were fully restored (Figure 125). Removing the dam enables salmon to access 17 additional river miles of spawning and rearing habitat (EPA, 1997). Data are from the remove Dwinnell Dam RMS simulations, and have been averaged to weekly values (Null, 2008).

In reality, removing Dwinnell Dam would largely restore the natural hydrograph with flood pulses, increase baseflow through summer, and would improve gravel recruitment, a habitat criteria not considered here (NRC, 2004). Cost estimates for removing Dwinnell Dam or similar sized earthen dams could not be found, so removal costs are assumed to be $15 million, not including water replacement costs or lost agricultural value (Table 19). Water replacement costs are significant, there are approximately 15,500 irrigable acres in MWCD. At $1000/acre (a low value), water replacement could be at least an additional $15 million. Removing Dwinnell Dam is modeled as a binary integer variable, so the only option is to remove the dam in its entirety. Because cost estimates for removing Dwinnell Dam were unavailable, this alternative is included here as an academic exercise.

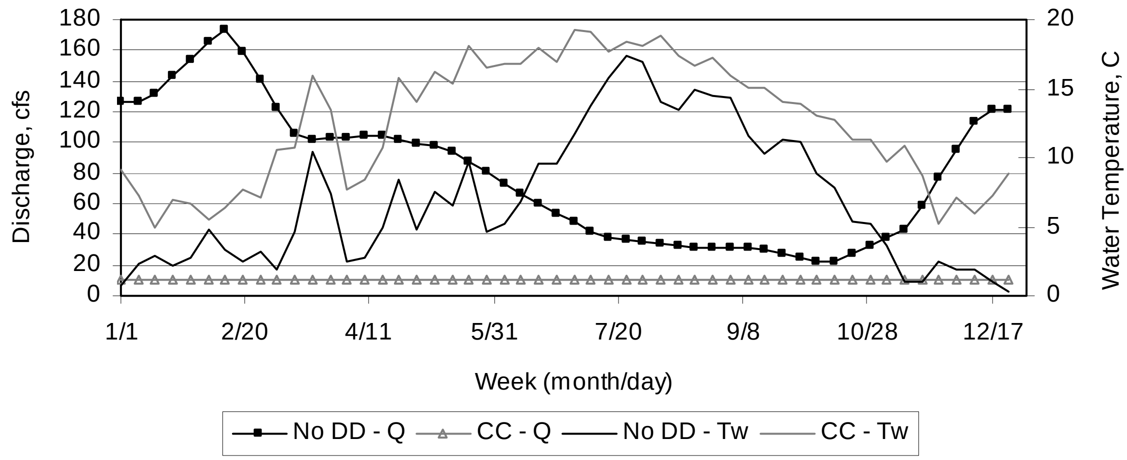


Figure 125. Flow and water temperature at Dwinnell damsite with current conditions (CC), and removing Dwinnell Dam (No DD)

Results

Model results help quantify tradeoffs of increasing habitat capacity for native salmon in the Shasta River, and aid decision making regarding restoration of instream habitat. Results should be interpreted not in absolute numbers of fish as there is little escapement data to test the model, but rather by relative numbers or percentage change.

Under current conditions (no restoration options), 99 fish out-migrate from the Shasta River. Week 32 (Aug. 6 – Aug 12) creates a bottleneck in the juvenile rearing stage, which limits coho escapement (Figure 126). Alevin continue to enter the juvenile life stage through the end of June, so through June there is considerable flexibility in the model. However, habitat conditions worsen in July and August, and juvenile fish must rear for a year in the river. Results show a die back from over 1400 juvenile fish to 99 fish, a reduction of nearly 93%. The remaining 99 fish move primarily between reach 2 and the Big Springs reach, where flow and temperature conditions are amenable to juvenile coho. Week 32 has warmer thermal conditions than surrounding weeks, with temperatures well above 20ºC (Figure 127). Flow is lower than neighboring weeks in reaches 7 – 10.

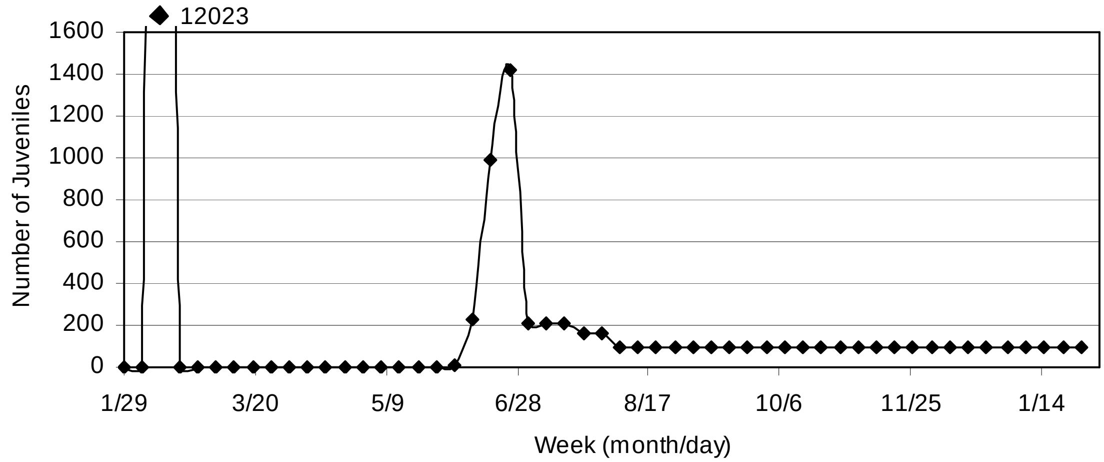


Figure 126. Total juvenile rearing in all reaches with current conditions

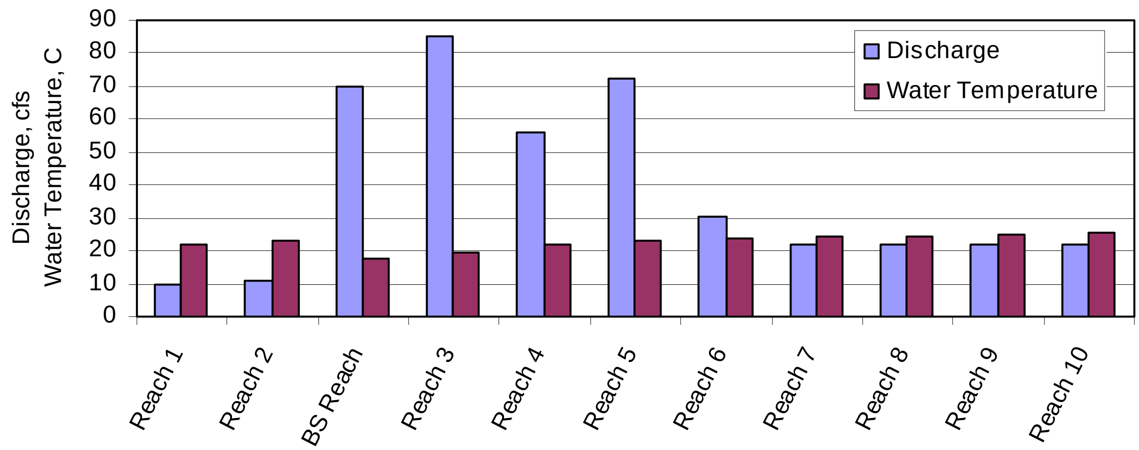


Figure 127. Week 32 flow and temperature current conditions

There is slight improvement in coho escapement by adding flow and riparian shading to the river (Figure 128). When $227,000 has been spent on increasing riparian vegetation, supplemented with approximately 56.5 cfs-yr of additional flow, escapement rises from 99 to 138 smolts, a 39% increase. Out-migration then plateaus until the restoration budget increases to $15,000,000. Removing Dwinnell Dam raises out-migration to 566 smolts. However, restoring Big Springs Creek also costs $15,000,000 and has a much greater benefit, with 1876 smolts. Maximum escapement is 2466 fish when Big Springs Creek is restored, approximately 1269 cfs-yr is added (sum of all weeks and reaches), and all reaches are fully shaded. When escapement is 2466, removing Dwinnell Dam and relocating the GID diversion have no effect.

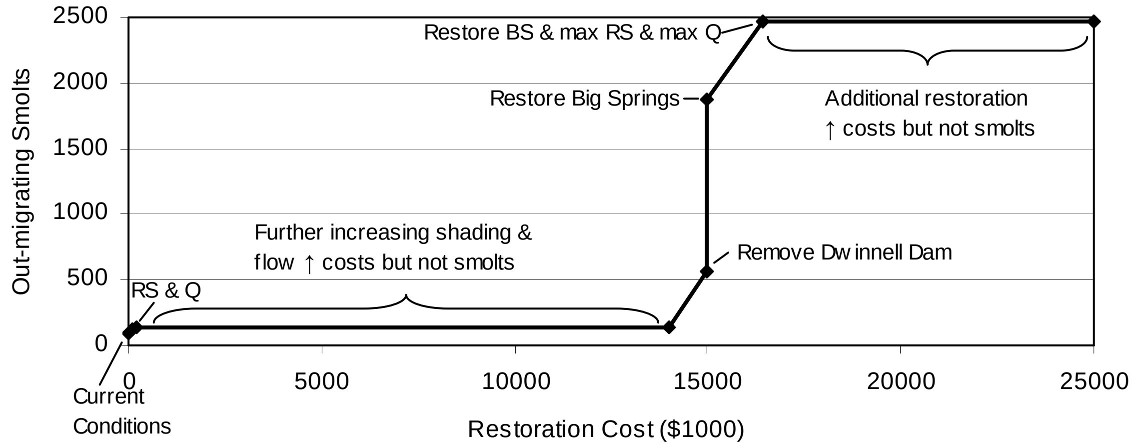


Figure 128. Restoration Tradeoff Curve

Additional Flow

Flow is added to each reach to improve instream conditions, except when the restoration budget is $0 or $15,000,000, when restoring Big Springs Creek or removing Dwinnell Dam most enhance instream conditions (Figure 129). When Big Spring Creek has not been restored or Dwinnell Dam removed, maximum annual additional flow volume to the Shasta River can reach 8933 cfs over the course of a year (totaling all reaches and weeks) (Figure 130), although no benefit occurs to out-migrating smolts beyond 56.5 cfs-yr of additional flow. It is consistently optimal to add the most flow to reach 1 (which contributes to downstream reaches) (Figure 130). After Big Springs Creek has been restored, the maximum yearly flow is 1269 cfs-yr because flow is limited by upper bounds equal to unimpaired conditions.

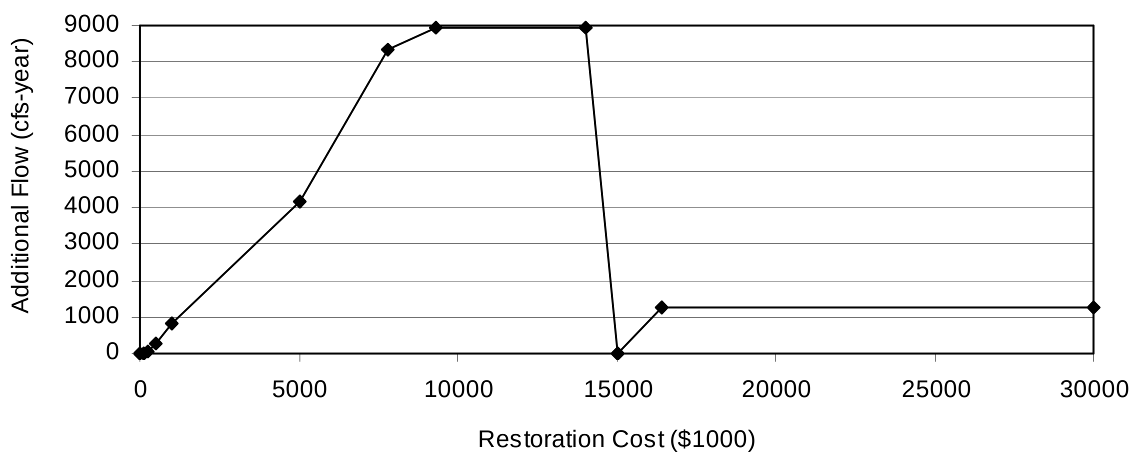


Figure 129. Additional flow tradeoff curve

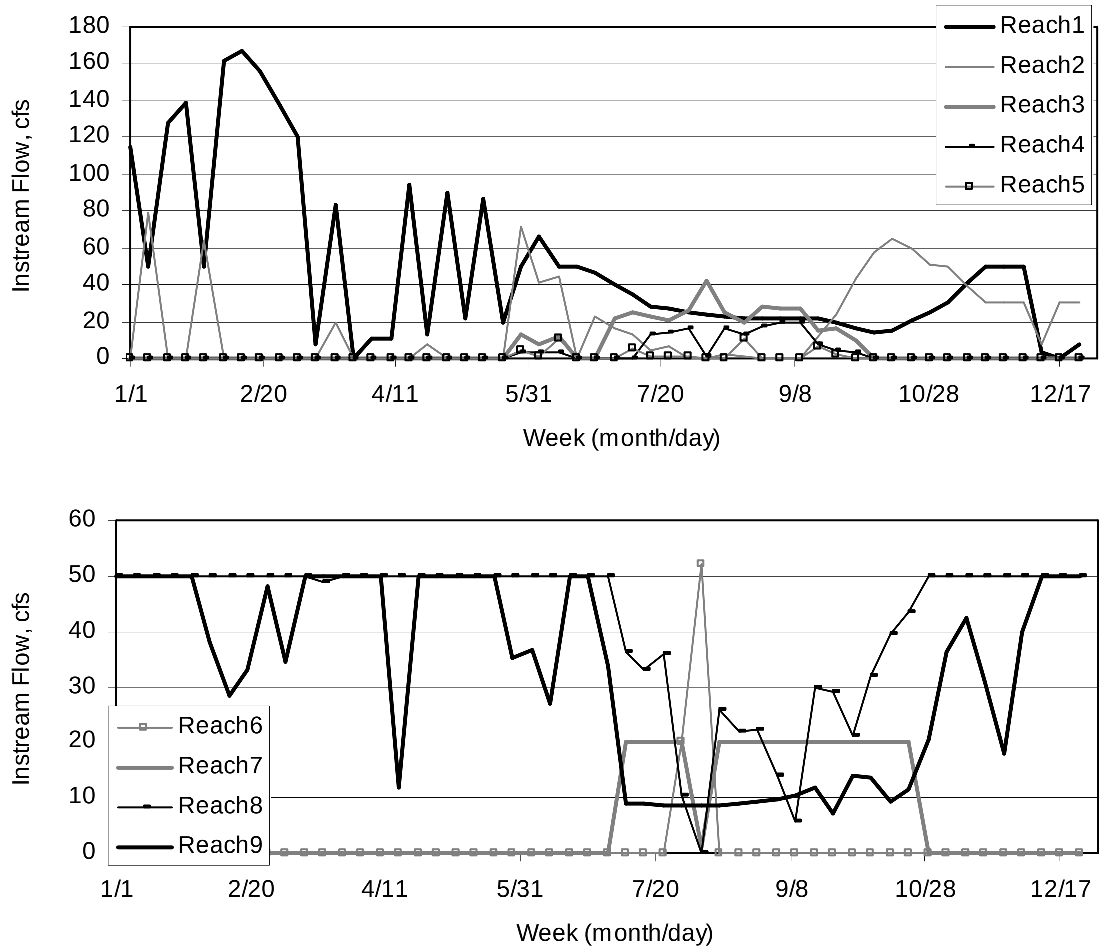


Figure 130. Additional Flow by reach (restoration budget = $10 million)

The model is fairly insensitive to changes in flow, in part due to the logistic habitat surfaces used here. Given uncertainty regarding the relationship between instream flow, water temperature, and fish survivorship, the logistic habitat surfaces developed are academic for this analysis. Detailed studies would be required to develop more robust curves. It is also likely that adding more flow at warm temperatures to the Shasta River may have a negligible effect on coho productivity.

Increased Riparian Shading

When budgets are limiting, the model always opts to shade the upper reaches first (Figure 131). Reaches 8 and 9 are always the last for increased riparian shading to be optimal, because in those reaches increased riparian shading makes only a small difference in heating rates (Figure 123). Although 100% is considered fully shaded for the Shasta River, dual values on the upper bound constraint are useful to show for which reaches it is particularly valuable to maintain cool water temperature (Figure 132). Maintaining cool thermal conditions in reach 1 below Dwinnell Dam is most beneficial for fish production with restoration budgets under $15 million. When Big Springs has been restored, maintaining water temperature through reach 4 also becomes a priority because cool inflow has been added to reach 3.

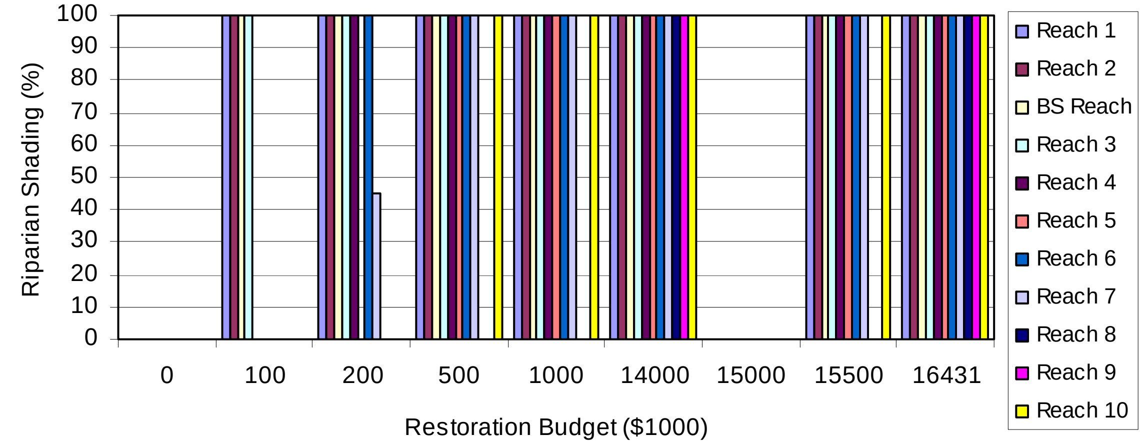


Figure 131. Percentage of riparian shading by reach and restoration budget

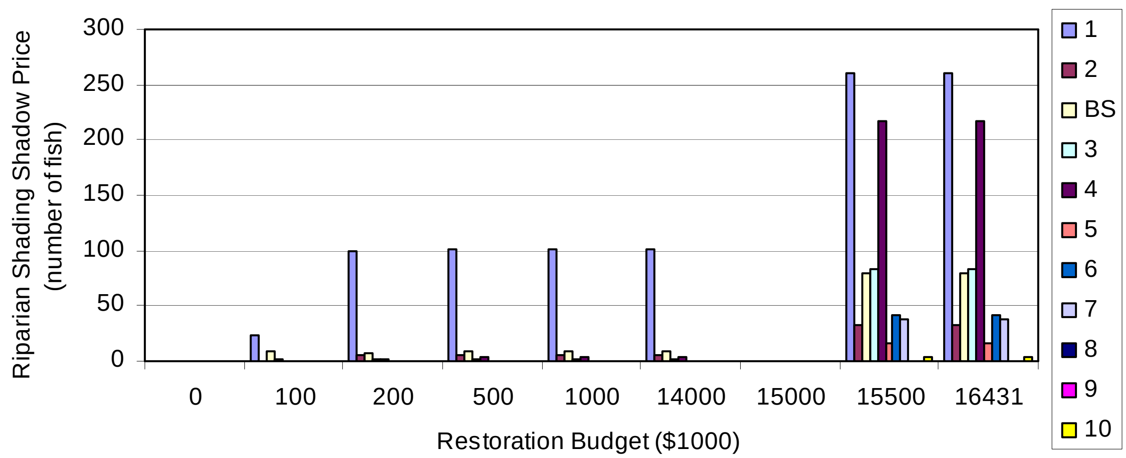


Figure 132. Shadow fish of additional riparian shading

Relocating GID

Relocating the GID diversion has little benefit to fish productivity. Using $1,000,000 to move GID results in 100 smolts, (1 more fish than current conditions), while spending the same amount on increasing instream flow and riparian vegetation results in 138 smolts. Again, this model does not explicitly model heating, but rather uses a mass balance approach for the heat budget and different water temperature initial and boundary conditions for various restoration alternatives. The results obtained from simulation modeling show this option may reduce water temperature for approximately 1ºC over approximately 15 river miles, although it did not lead to appreciable improvement for fish here.

Restoring Big Springs Creek

As stated above, when Big Springs Creek is restored, escapement rises to 1876 fish, a 1795% increase from current conditions. Week 32 still creates a bottleneck in juvenile rearing when Big Springs is restored (Figure 133). This implies that meteorological conditions will continue to create bottlenecks in weeks or life stages even with restoration, although the effects will not be as dire. When Big Springs Creek is restored, Big Springs and the canyon reach (reach 10), have the most alevin (recall that no spawning occurs in reaches 6 – 9 or above Dwinnell Dam (ADD) since Dwinnell Dam has not been removed) (Figure 134).

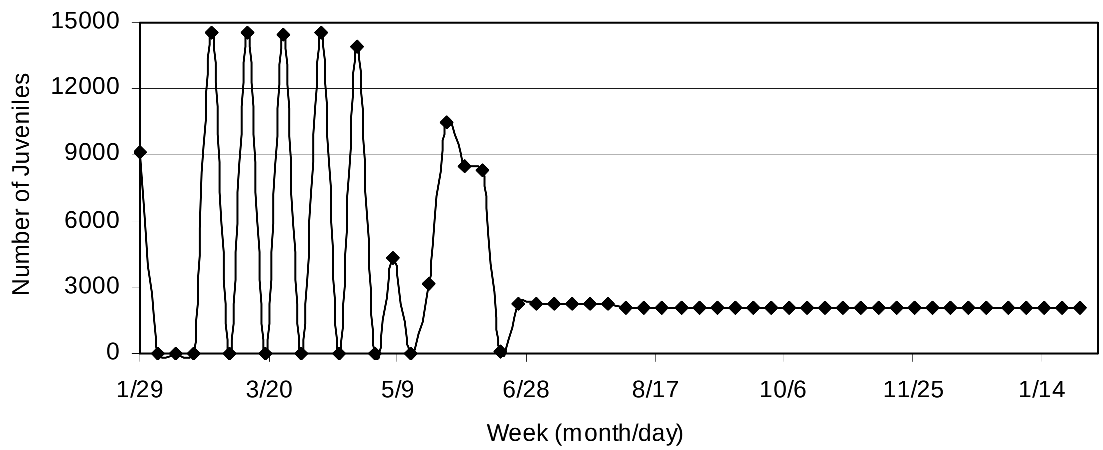


Figure 133. Total juvenile rearing in all reaches when Big Springs Creek is restored

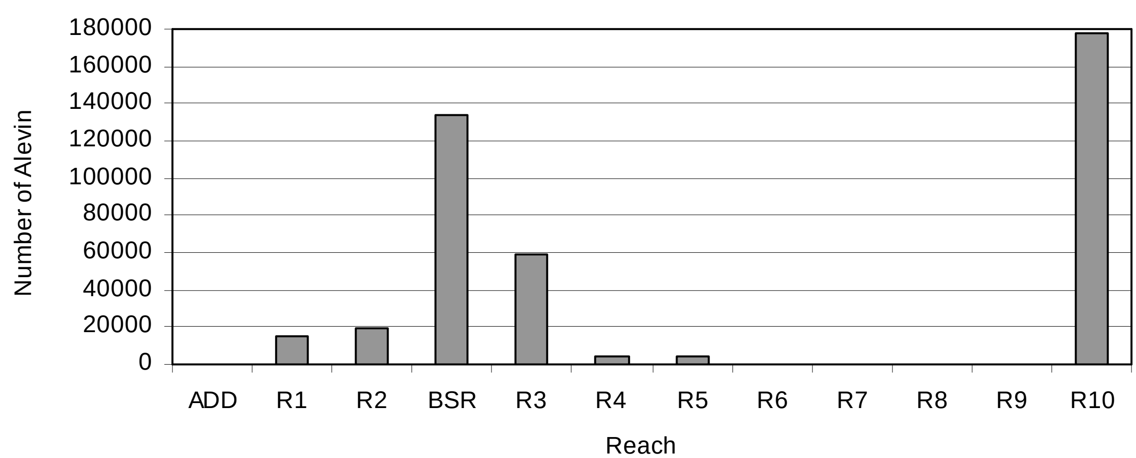


Figure 134. Total alevin by reach when Big Springs Creek is restored

Removing Dwinnell Dam

Removing Dwinnell Dam benefits coho production in the Shasta River, although not as much as restoring Big Springs Creek. Most likely, either Big Springs would be restored or Dwinnell Dam removed, depending on politics, public support, and institutional agreements. When Big Springs Creek has already been restored, there is little benefit to removing Dwinnell Dam in this model (although in reality, removing the dam would enable fish to access an additional 17 miles of habitat and restore a natural hydrograph which are not considered here). For this study removing Dwinnell Dam would only be worthwhile if it were substantially cheaper than restoring Big Springs Creek. Removing Dwinnell Dam could cost more than was assumed for this study, and sensitivity analysis on cost estimates would be a useful future exercise.

Limitations

Limitations of this study are divided into limitations of the modeling approach, and data limitations for the Shasta River case study. Modeling limitations include managing non-linear solution space, greatly simplified fish ecology, no representation of physical processes, considering only flow and water temperature as factors limiting coho production, and coarse spatial and temporal resolution. Data limitations include cost estimates, lack of tailwater data, and only one year of estimated flow and water temperature data.

Nonlinear models are inherently more difficult to solve that linear ones. Nonlinear models do not guarantee global optima when convexity has not been proven, as in the fish habitat model presented here. Thus, populating adjustable cells in the model with initial values that lead to global solutions is necessary. Otherwise, local optima would often be returned (in which the solution is best only in the immediate neighborhood).

Fish ecology is greatly simplified here. In reality, fish population dynamics are complicated, and many details remain unknown. Habitat logistic surfaces could be improved, which would most likely make the model more sensitive to small changes in flow and water temperature. All instream habitat parameters except flow and water temperature have been ignored, although other water quality concerns, barriers to migration, substrate, and Klamath River conditions contribute to salmon decline (DWR, 2001; NRC, 2004).

The model uses a mass balance approach for water and heat budgets to estimate instream conditions in all reaches. Meteorological and geomorphology processes are absent in the model. Thus changes to flow which also change the relative surface area at the air-water interface and travel time should alter water temperature, but do not here.

The coarse temporal scale of the model eliminates many important fish habitat criteria such as maximum daily water temperature, duration of elevated temperature, and daily minimum water temperature. Furthermore, the river below Dwinnell Dam has reaches up to 17 river miles. Finer model resolution would improve results. Data exists to improve model resolution, but model run times would increase. Additional data must be gathered to model flow and thermal changes in two dimensions.

Much of the data used for this application to the Shasta River could be improved. Refinement of cost estimates would lead to more certain results. Improved input data on the thermal effects of tailwater returns would allow tailwater management to be a restoration decision variable in the model. Finally only one year is modeled here. More flow and temperature data (currently being collected by U.C. Davis, Watercourse Engineering, Inc, CDWR, and NCRWQCB) would aid understanding of the system, as well as increase knowledge about how habitat and fish populations are affected by different water years, meteorological conditions, and fish cohorts.

Future work includes incorporating additional fish species to understand the effects of competition (for habitat) and avoid single species management. Additionally, incorporating physical processes would increase the utility of this type of model, so that the thermal affects of greater volumes of water could be assessed. Finally, problem linearization would improve model run times and human involvement because populating the model with practical initial values would not be necessary.

Discussion

The optimization model described here illustrates an approach to compare habitat improvement for one fish species by linking restoration actions with fish habitat and economic costs. This research shows the relative value of different restoration activities for fish productivity in the Shasta Valley, providing one tool for local stakeholders and decision makers to weigh decisions and justify (or eliminate) costs of restoration in the Shasta Valley. Results from this approach illustrate the benefit to fish from each restoration activity, as well as the quantity of water reduced from other uses, and associated costs. This allows both environmental water use efficiency and the economic efficiency of restoration decisions to be measured by fish habitat. It also quantifies impacts to current Shasta Valley water by estimating water allocations necessary for restoration as well as associated costs.

Of the restoration actions evaluated, restoring Big Springs Creek provides the most improvement for fish habitat, increasing smolts by 1795% (Table 23). Removing Dwinnell Dam improves escapement by 472%, a significant increase, although minor when compared to restoring Big Springs Creek. Increasing riparian shading benefits raises the number of fish out-migrating from the Shasta River by 30.3%. Simple flow changes such as increasing flow or relocation the GID diversion provide negligible benefit to fish habitat, although the optimization model used here is not driven by physical processes, so increasing volume or decreasing the air-water interface cannot be analyzed.

Table 23. Smolt production, flow increase, and cost of restoration alternatives

	Number of Out-migrating Smolts	Additional Environmental Flow (cfs)	Cost ($)
Current Conditions	99	0	0
Additional Flow	101	55.56*	50,000
Full Riparian Shading	129	0	235,000
Relocate GID Diversion	100	0	1,000,000
Restore Big Springs Creek	1876	1896	15,000,000
Remove Dwinnell Dam	566	3610	15,000,000

* Additional flow does not increase smolt production

These results suggest increasing instream flow without improving water quality results has little benefit to fish habitat. Evaluating the extent to which additional flow will enhance instream conditions may eliminate water intensive restoration decisions, resulting in greater environmental water use efficiency. Here additional flow is of little value to fish production unless water temperature is also reduced. Additional findings include:

- Restoration alternatives can be ranked in terms of value to fish habitat.
    - o Restoring Big Springs provides the most benefit, but removing Dwinnell Dam is a good second choice.

- Systems analysis is most useful when paired with simulation models that incorporate greater detail and explicitly model physical processes.
    - o The heat budget approach used here shows little value in relocating the GID diversion, although simulation modeling shows thermal benefit from moving the diversion.

- Improving water quality (rather than increasing quantity) is beneficial for fish.
    - o Increasing riparian shading to preserve cool water temperatures is valuable for fish productivity.

- Bottlenecks in the life history of fish still occur when restoration activities have improved instream conditions, although the consequences are less severe.
    - o Restoration could provide a buffer against poor ocean conditions or possible habitat degradation associated with climate change.

- The tradeoff curve between economic costs of restoration and number of out-migrating smolts is not smooth, some alternatives are corner points that result in large increases in cost or fish productivity.

- Modeling suggests substantial investment in fish habitat must occur before escapement increases in the Shasta River.

- Fish productivity has an upper bound, at which point additional water temperature habitat enhancement measures have no value.
    - o Another habitat factor may then be limiting fish production, which should then be the focus of restoration activities.

- Systems analysis provides a helpful approach for managing ecosystems, as well as traditional water uses.

References

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51:1389-1406.

CBSED (City and Borough of Sitka Electric Department). 2005. Draft Habitat Suitability Curves for Coho Salmon and Steelhead Trout, Sawmill Creek Alaska. Prepared for Blue Lake Hydroelectric Project, FERC No. 2230, Relicensing.

Cohon, J.L. 1978. Multiobjective Programming and Planning. Academic Press. San Francisco, CA. 333 pp.

CDFG (CA. Dept. of Fish and Game). 2002. Status Review of California Coho Salmon North of San Francisco. Report to the California Fish and Game Commission. Available online: http://www.dfg.ca.gov/fish/Resources/Coho/SAL_CohoStatusRpt.asp. Accessed 6/2008.

CDFG (CA. Dept. of Fish and Game). 2003. Shasta and Scott River Pilot Program for Coho Salmon Recovery: with recommendations relating to Agriculture and Agricultural Water Use. Prepared by the Shasta-Scott Coho Salmon Recovery Team.

CDFG (CA. Dept. of Fish and Game). 2003b. Shasta River Fish Counting Facility, Chinook and Coho Salmon Observations in 2002, Siskiyou County, CA. Available online: http://www.krisweb.com/biblio/shasta_cdfg_hampton_2003_2002count.pdf. Accessed 7/2008.

CDWR (CA. Dept. of Water Resources). 2001. Watershed Study Plan: Water Quality and Aquatic Habitat Characterization in the Shasta River. Prepared by G.L. Boles. Northern District, Red Bluff, CA.

Deas, M.L., A. Abbott, A. Bale. 2003. Shasta River Flow and Temperature Modeling Project. Prepared for Watercourse Engineering, Inc. Sponsored by the Shasta Valley Resource Conservation District with funding by CDFG from the CA. Coastal Salmon Restoration Program. Napa, CA.

Draper, A.J., M.W. Jenkins, K.W. Kirby, J.R. Lund, R.E. Howitt. 2003. Economic-Engineering Optimization for California Water Management. Journal of Water Resources Planning and Management. 129 (3): 155-164

EPA (U.S. Environmental Protection Agency). 1997. Designated Critical Habitat; Central California Coast and Southern Oregon/Northern California Coast Coho Salmon. Federal Register: 62 (227). Last updated 3/3/2008. Available online: http://www.epa.gov/fedrgstr/EPA-SPECIES/1997/November/Day-25/e30865.htm. Accessed 6/2008.

Higgins, J.M., W.G. Brock. 1999. Overview of Reservoir Release Improvements at 20 TVA Dams. Journal of Energy Engineering 125(1): 1- 17.

Hillier, Y., G. Lieberman. 1967. <u>Introduction to Operations Research</u>. Holden-Day Publishers. San Francisco, CA. 639 pp.

Homa, E.S., R.M. Vogel, M.P. Smith, C.D. Apse, A. Huber-Lee, J. Sieber. 2005. An Optimization Approach for Balancing Human and Ecological Flow Needs. Proceedings of the EWRI 2005 World Water and Environmental Resources Congress, ASCE, Anchorage, AK.

Jeffres, C. (U.C. Davis Watershed Sciences Center). Personal communication. 4/22/2008.

Jeffres, C., E. Buckland, J. Kiernan, A. King, A. Nichols, S. Null, J. Mount, P. Moyle, M. Deas,. 2008. Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California – Water Year 2007. Prepared for the California Nature Conservancy by U.C Davis Center for Watershed Sciences and Watercourse Engineering, Inc.

Jowett, I.G. 1997. Instream Flow Methods: A Comparison of Approaches. Regulated Rivers: Research & Management 13: 115 – 127.

Labadie, J.W. 2004. Optimal Operation of Multireservoir Systems: State-of-the-Art Review. Journal of Water Resources Planning and Management 130(2): 93-111.

Liebman, J.C., W.R. Lynn. 1966. The Optimal Allocation of Stream Dissolved Oxygen. Water Resources Research 2(3): 581-591.

Lindo Systems Inc. 2005. What's Best! Spreadsheet Solver. Chicago, IL. Website: www.lindo.com

Loomis, J.B. 2000. Environmental Valuation Techniques in Water Resources Decisions Making. Journal of Water Resources Planning and Management 126(6): 339 – 344.

Martin, J.L., S.C. McCutcheon. 1999. <u>Hydrodynamics and transport for water quality modeling</u>. Lewis Publishers, New York. 794 pp.

Moyle, P.B. 2002. <u>Inland Fishes of California</u>. University of California Press. Berkeley and Los Angeles, California.

National Resources Council (NRC). 2004. <u>Endangered and Threatened Fishes in the Klamath River Basin: Causes of Decline and Strategies for Recovery</u>. Committee on Endangered and Threatened Fishes in the Klamath River Basin, National Research Council. The National Academy Press.

Nickelson, T.E., J.D. Rodgers, S.L. Johnson, M.F. Solazzi. 1992. Seasonal Changes in Habitat Use by Juvenile Coho Salmon (Oncorhynchus kisutch) in Oregon Coastal Streams. Canadian Journal of Fisheries and Aquatic Sciences 49:783-789.

NCRWQCB (CA. North Coast Regional Water Quality Control Board). 2006. Staff Report for the Action Plan for the Shasta River Watershed Temperature and Dissolved Oxygen Total Maximum Daily Loads. Santa Rosa, CA. Available online:http://www.swrcb.ca.gov/northcoast/water_issues/programs/tmdls/shasta_river/staff_report.shtml. Accessed: 6/2008.

Null, S.E., J.R. Lund. 2006. Restoring Hetch Hetchy Valley: The Role of Modeling in Policy. Eos, Transactions, American Geophysical Union 87(42): 449 – 451.

Null, S. 2008. Improving managed Environmental Water Use Efficiency: Shasta River Flow and Temperature Modeling. U.C. Davis PhD Dissertation. Geography Graduate Group.

Palmer, R.N., R.M. Snyder. 1985. Effects of Instream Flow Requirements on Water Supply Reliability. Water Resources Research 21(4): 439 – 446.

Quinn, J.M., P.M. Brown, W. Boyce, S. Mackay, A. Taylor, T. Fenton. 2001. Riparian Zone Classification for Management of Stream Water Quality and Ecosystem Health. Journal of the American Water Resources Association 37(6): 1509-1515.

Richter, B.D., J.V. Baumgartner, R. Wigington, D.P. Braun. 1997. How much water does a river need? Freshwater Biology 37: 231 – 249.

Sale, M.J., E.D., Brill, Jr., E.E. Herricks. 1982. An Approach to Optimizing Reservoir Operation for Downstream Aquatic Resources. Water Res. Research 18(4): 705–712.

TNC (The Nature Conservancy). 2005. A Strategy for Enhancing Flow and Protecting Anadromous Fish Habitat in the Shasta River. Prepared by Kat Ridolfi.

Vignola, E., M. Deas. 2005. Lake Shasta Limnology. Prepared for the Information Center for the Environment and the North Coast Regional Water Quality Control Board by Watercourse Engineering, Inc. Davis, CA.

Watanabe, M., R.M. Adams, J. Wu. 2006. Economics of Environmental Management in a Spatially Heterogeneous River Basin. American Journal of Agricultural Economics 88(3): 617 – 631.

Welsh, Jr., H.H., G.R. Hodgson, B.C. Harvey. 2001. Distribution of Juvenile Coho Salmon in Relation to Water Temperatures in Tributaries of the Mattole River, California. North American Journal of Fisheries Management 21:464-470.

Williams, J.C., C.S. ReVelle, D.J. Bain. 2003. A decision model for selecting protected habitat areas within migratory flyways. Socio-Economic Planning Sciences 37(2003): 239 – 268.

Chapter 7 – Discussion and Conclusions

This dissertation studies environmental water use and methods to aid restoration decisions using multiple scales and approaches, including theoretical analysis, field monitoring, simulation, and optimization. This chapter summarizes the major points and findings of each of the studies of this dissertation, with emphasis on how each section relates to environmental water use efficiency (WUE), the original contributions of the research, how each component relates to the approaches used in other chapters, and relevant future work. The different approaches and scales contribute to improving understanding of environmental water use and habitat enhancement. This chapter ends with final remarks on environmental WUE as a management strategy.

Environmental Water Use Efficiency Theory

The initial chapters explore the idea of improving environmental WUE, much as the urban and agricultural sectors have improved their water use efficiencies. Maximizing the environmental benefit for a given amount of water dedicated to environmental uses is a relatively new concept (Begley et al., 2006; Lankford, 2003), but may prove worthwhile to save instream habitats, species, money, and time. Managed environmental WUE maximizes environmental benefits from a given quantity of water, and is related to the idea of restoration efficiency, or ensuring that restoration work achieves its goals. Environmental WUE can be as simple as figuring out how much water is needed to meet a specific goal, such as how much water is necessary to mobilize gravels. In regulated systems, only that much water should be released, saving excess water (if it exists) for other environmental goals. This approach implies that environmental systems must be managed more pro-actively for the long term, which is different from many short-term crisis management projects common in aquatic restoration today.

The new contributions of chapters 1 and 2 are to further the concept of actively managing environmental water to maintain aquatic and riparian habitat, and explore methods of evaluating restoration alternatives. These chapters critically evaluate the potential of environmental water management, beginning with discussion of the idea and development of illustrative models for environmental and restoration efficiency. The work undertaken here focuses on improving instream conditions for native fish, although that is only one potential application for environmental WUE. Other sections of this dissertation use the environmental WUE framework presented in the initial chapters to evaluate solutions to practical problems, such as dwindling fish populations caused by high water temperatures and low-flow conditions in California's Shasta River.

Field Monitoring

Field monitoring on the Shasta River lasted one year and included longitudinal and lateral temperature analysis, thermal diversity monitoring, and spring inflow monitoring at The Nature Conservancy's Nelson Ranch. In addition, agricultural tailwater return monitoring was conducted at Nelson and Meamber Ranches. Field monitoring is instrumental for understanding current conditions and problems facing natural systems. Furthermore, it provides data needed to calibrate and test models. This monitoring data is much more detailed than that produced by numerical models, making small-scale trends apparent that at this time cannot be reproduced with computer models. For this reason, monitoring is typically an ongoing process throughout restoration, and understanding the efficacy of restoration projects is impossible without field monitoring. Conversely, modeling helps provide a unifying and synthesizing framework for field data collection.

New findings were discovered where thermal conditions and instream flow were monitored on the Shasta River, helping to characterize current conditions and assess factors limiting salmonid survival. Water temperature differed between the upstream and downstream property boundaries, with maximum water temperature at the downstream boundary occurring at night. This implies that warm water is being inherited from upstream of Nelson Ranch, combined with atmospheric heating. Lateral thermal variability occurs during summer at river margins, but is largely absent during other seasons. Small potential local thermal refugia were found during summer, where temperatures are 1-2ºC cooler than surrounding water, although the sources of the cool water were not examined. Agricultural tailwater and springfed tributaries had higher thermal diversity than the mainstem Shasta River, and could be a source of warmer or cooler water depending on season and time of day. Cumulative effects of these small inflows were not studied.

The water temperature data collected on the Shasta River was used to fill in data gaps because most temperature data has been collected only during summer months. Regardless, many information gaps remain. Monitoring is the basis for most other types of studies, such as modeling, and should continue to be a research priority. Flow and temperature data from additional locations on the Shasta River is currently being collected and will be most useful to understand travel time of the river and to test hydrodynamic and water quality models. Also, more research is needed to understand the cumulative effects of warm-water inflows over the length of the Shasta River, such as agricultural tailwater returns. Better accounting for water and thermal energy, through channelized, overland, and subsurface flow is needed. In general the thermal effects of cumulative tailwater returns to the Shasta River are poorly understood, but may substantially affect the thermal regime of the Shasta River. Finally, all temperature data should be analyzed in tandem with fish counts and other physical and biological habitat data to get a better understanding of instream conditions and the factors limiting fish survival (Jeffres et al, 2008).

Shasta River Simulation Modeling

Water temperature and instream flow were simulated for the Shasta River in 2001 using RMSv4, an hourly, one dimensional hydrodynamic and water quality simulation model. Although other modeling studies have been done on the Shasta River (Abbott, 2002; Deas et al., 2003; Geisler, 2005), this research differs because modeling was done for an entire year to assess conditions through all seasons, and the worst conditions of the year could be pinpointed for all model runs. Also many model runs were done for conditions not previously simulated, such as relocation of the GID diversion, restoring Big Springs Creek, removing Dwinnell Dam, and unimpaired conditions. Other model runs, such as return flow analysis on Nelson Ranch, provide rough estimates of the thermal effects and flow changes from agricultural return flow, and are useful to improve understanding of how much return flow can be added to the Shasta River before mainstem water temperature is affected.

Results suggest options exist to improve instream flow and temperature conditions on the Shasta River, but cool water is needed in the upper reaches for restoration to be effective, and a mix of alternatives may provide the most benefit. Different restoration alternatives target different river reaches, so water managers should work with fish biologists and stakeholders for restoration to be most effective. Restoring Big Springs Creek shows particular promise, as it maintains cool water temperatures through much of the Shasta River.

Findings from the simulation modeling completed on the Shasta River are pertinent to environmental WUE. First, substituting higher water quality can benefit native species without increasing environmental water allocations, where water quality is limiting. Also, modeling as an exercise helps researchers understand aquatic systems, as many components are considered in building the model. Thus, it becomes easier to understand how flow changes affect instream habitat and how much water is necessary (and when it is needed) to meet restoration objectives. Water temperature was examined in this case, although similar models could be used to analyze geomorphic conditions, dissolved oxygen, vegetation recruitment, physical migrations, or other physical or biological conditions. If restoration objectives are well-defined, modeling studies are more effective because restoration actions can be scrutinized for how well they meet objectives. Finally, modeling is a good method of identifying data gaps. Thus, modeling and field monitoring are closely related, and often are carried out in conjunction.

Future work for simulating the Shasta River includes using a two-dimensional model to gain insight into lateral thermal variability over the length of the Shasta River, and explicitly modeling tributaries. Big Springs Creek may be instrumental in restoring habitat conditions in the Shasta River, but is modeled as a boundary condition in the work undertaken here. At the time of this research, little flow and temperature data had been collected on Big Springs Creek. However, more data exists now and continues to be collected (U.C. Davis; Watercourse Engineering, Inc.; Null, unpublished data) which could be incorporated into a model to better understand how restoring Big Springs Creek would affect the Shasta River.

Systems Analysis for Environmental Water Management

Optimization modeling was used to evaluate which restoration options provide the most habitat improvement within a restoration budget. The approach was especially fitting for environmental WUE because it quantifies instream habitat and ranks restoration alternatives, providing an example to manage environmental water and budget allocations efficiently. The optimization model created allows restoration actions to alter instream flow and water temperature, which in turn affects the number of fish of different age classes that survive in a river (the model used Shasta River data and was based on the life history of coho salmon).

Like the Shasta River simulations, the optimization model showed that improving water quality could substitute somewhat for improving instream flow conditions. Bottlenecks in fish age classes still occurred with extensive restoration, but were not as critical. This has important implications because restoration could provide a buffer for events largely beyond human control, such as ocean conditions and global warming. The trade-off curve between fish production and economic costs of restoration was not smooth, rather there were corner-points where restoration actions result in large jumps in fish escapement or where costs increase with little increase in fish production. The modeling also demonstrates that substantial restoration investments must be made before fish production improves appreciably.

The optimization modeling developed here is directly applicable to improving environmental WUE. Through proposed restoration alternatives, the model links fish survival with additional water needed and associated costs, allowing insights in environmental WUE and restoration efficiency. Model results include the relative value of restoration actions, providing an example of systems analysis for informing ecosystem management. Finally, one of the more useful parts of optimization is that objectives and constraints must be explicitly defined, which may improve aquatic restoration activities and accountability of those activities. Although previous systems analysis research has included environmental objectives, the modeling completed here shows the approach could be applied to manage ecosystems as well as traditional water uses, and that it could help quantify restoration alternatives. This approach is interdisciplinary, merging methods from the geosciences, engineering, economics, and biology.

Optimization modeling could be improved by modeling more than one species of fish to avoid single species management. Additionally, model results could be improved by modeling more age classes of fish, such as in-migrants or spawners. Economic costs have no basis in some cases (such as removing Dwinnell Dam), and improved cost estimates would greatly improve model results and applicability. Incorporating physical processes, such as atmospheric heating, would also improve model results. Putting this work into a larger context, such as incorporating the value of a restored Shasta River to the Klamath River basin is a final idea for future systems analysis research to improve environmental conditions.

Discussion

Overall, the common theme of this dissertation is better managing environmental water allocations. To an extent this may include better managing traditional urban and agricultural water to allow for environmental enhancement, such as the case of specializing rivers for fish production or water supply. These chapters illustrate various methods and scales of improving aquatic habitat while considering traditional water uses, although many more exist, such as scheduled dam releases for pulse flows, environmental water banks, levee setbacks, or thermal curtains in reservoirs. In the future, increasing both understanding and implementation of managed natural systems when water is scarce may become more necessary and widespread, and innovative solutions and techniques will likely become more common. Further discussion and testing of environmental WUE, including possible applications, limitations, and implementation difficulties is warranted.

Conclusions

1) For the Shasta River, the flow and temperature benefits of restoring Big Springs Creek show great promise.
    - Cool water benefits exist over most of the mainstem Shasta River below Big Springs Creek during summer.

2) Cold water in the upper reaches of the Shasta River is necessary for any restoration approaches to be effective.

3) A combination of restoration alternatives most likely provides the most benefit for salmon habitat.

4) Substituting higher quality water can sometimes benefit native species without increasing environmental water allocations, when water quality is limiting species habitat.

5) Systems analysis in the forms of simulation and optimization modeling, backed by suitable field data collection, can provide insights for improving environmental management and efficient use of environmental water.

6) Environmental water use efficiency could potentially improve environmental performance and perhaps reduce some water management conflicts.

7) Environmental water use efficiency is an upcoming branch of water resources that merits additional study.

References

Abbott, A.G.P. 2002. The Effect of Riparian Vegetation on Stream Temperature in the Shasta River. Master's thesis. U.C. Davis Dept. of Civil and Environmental Engineering.

Begley, L., K. Edwards, S. Heard, L. Walling. 2006. Managed Environmental Water Use Efficiency: Evidence in Practice. Prepared for CDWR. U.C. Berkeley Goldman School of Public Policy. Berkeley, CA.

Deas, M.L., A. Abbott, A. Bale. 2003. Shasta River Flow and Temperature Modeling Project. Prepared for Watercourse Engineering, Inc. Sponsored by the Shasta Valley Resource Conservation District with funding by CDFG from the CA. Coastal Salmon Restoration Program. Napa, CA.

Geisler, L. 2005. Shasta River Flow, Temperature, and Dissolved Oxygen Model Calibration Technical Report. Prepared for ICE, UC Davis and NCRWQCB. UC Davis and Watercourse Engineering, Inc. Available online: http://www.waterboards.ca.gov/northcoast/programs/tmdl/shasta/060707/20AppendixDShastaRiverFlowTemperatureandDissolvedOxygenModelCalibrationTechnicalReport.pdf. Accessed 5/2007.

Jeffres, C., E. Buckland, J. Kiernan, A. King, A. Nichols, S. Null, J. Mount, P. Moyle, M. Deas,. 2008. Baseline Assessment of Salmonid Habitat and Aquatic Ecology of the Nelson Ranch, Shasta River, California – Water Year 2007. Prepared for the California Nature Conservancy by U.C Davis Center for Watershed Sciences and Watercourse Engineering, Inc.

Lankford, B.A. 2003. "Environmental Water Requirements: A Demand Management Perspective". Journal of the Chartered Institution of Water and Environmental Management 17(1): 19-22.

Appendix A – Additional Field Data

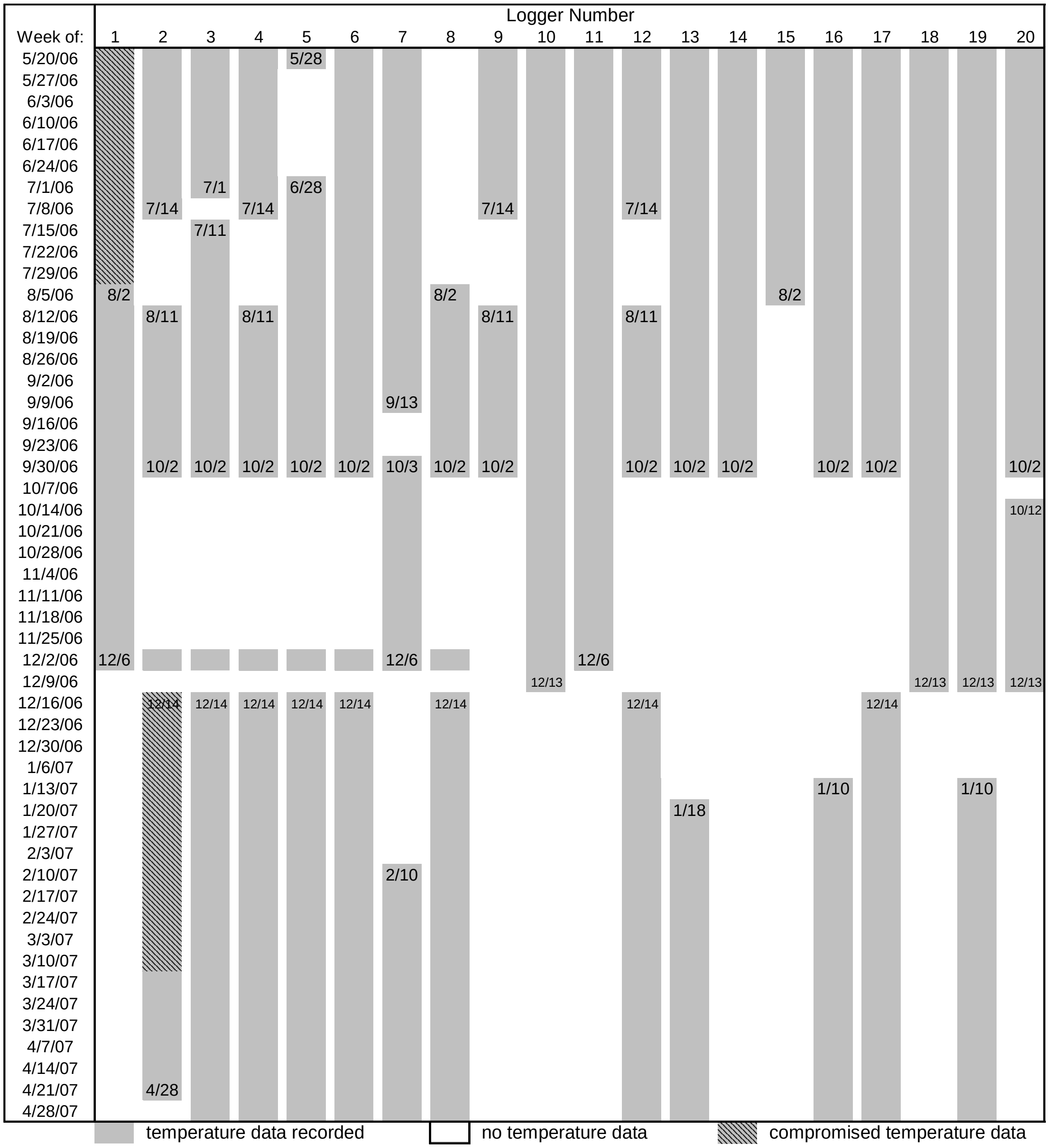


Figure 135. Nelson Ranch temperature logger completeness

168

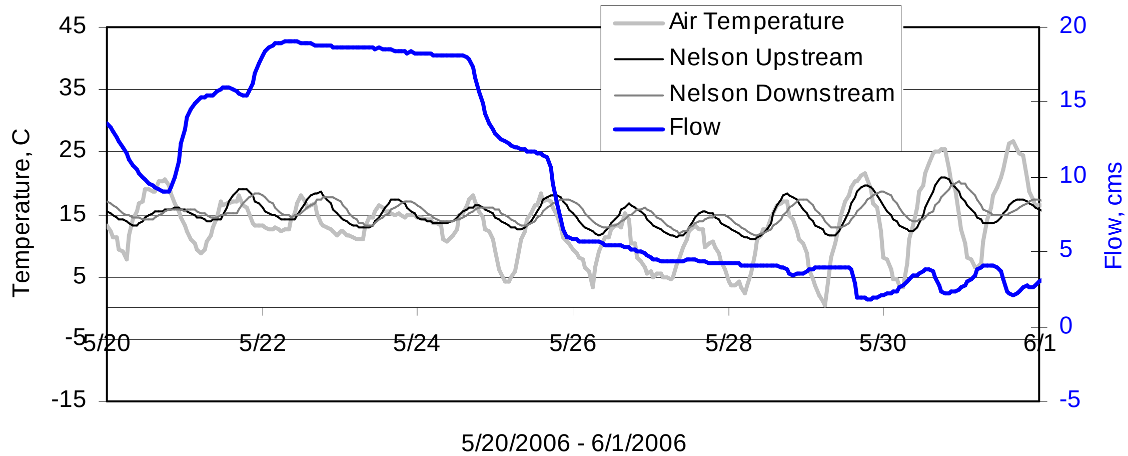


Figure 136. Water temperature, air temperature, and instream flow in the Shasta River; May, 2006

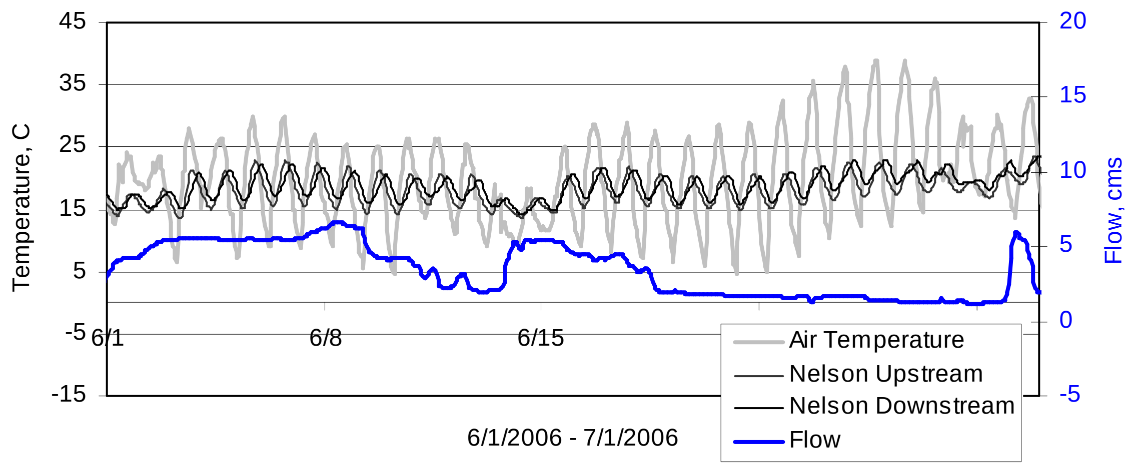


Figure 137. Water temperature, air temperature, and instream flow in the Shasta River; June, 2006

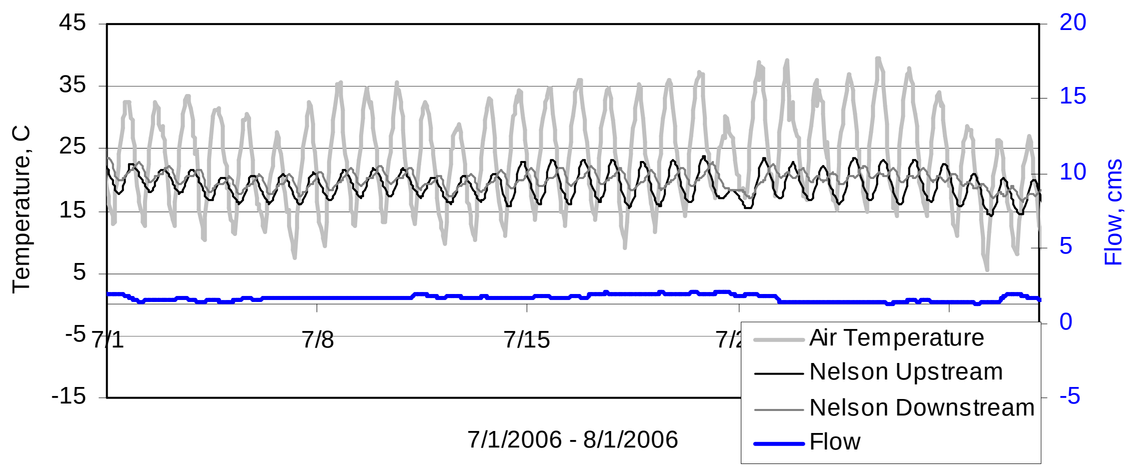


Figure 138. Water temperature, air temperature, and instream flow in the Shasta River; July, 2006

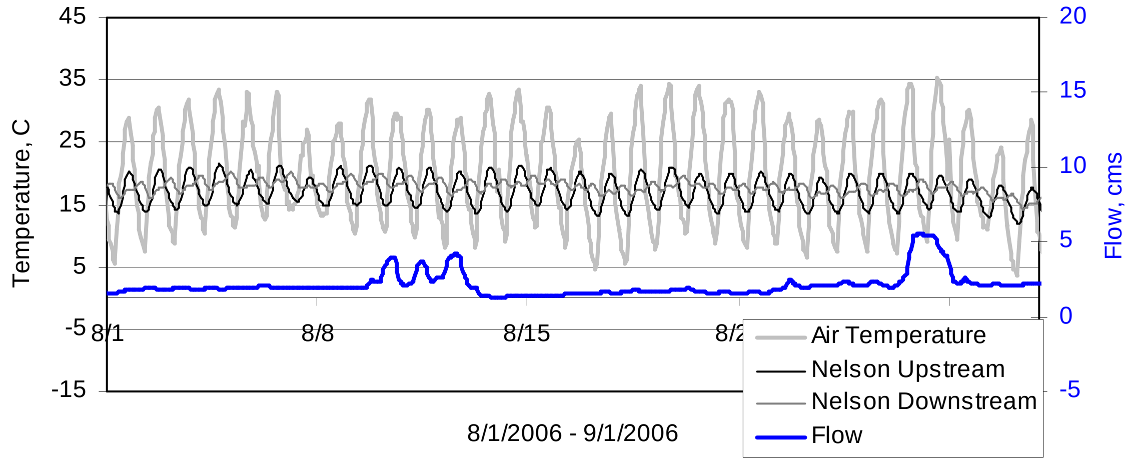


Figure 139. Water temperature, air temperature, and instream flow in the Shasta River; August, 2006

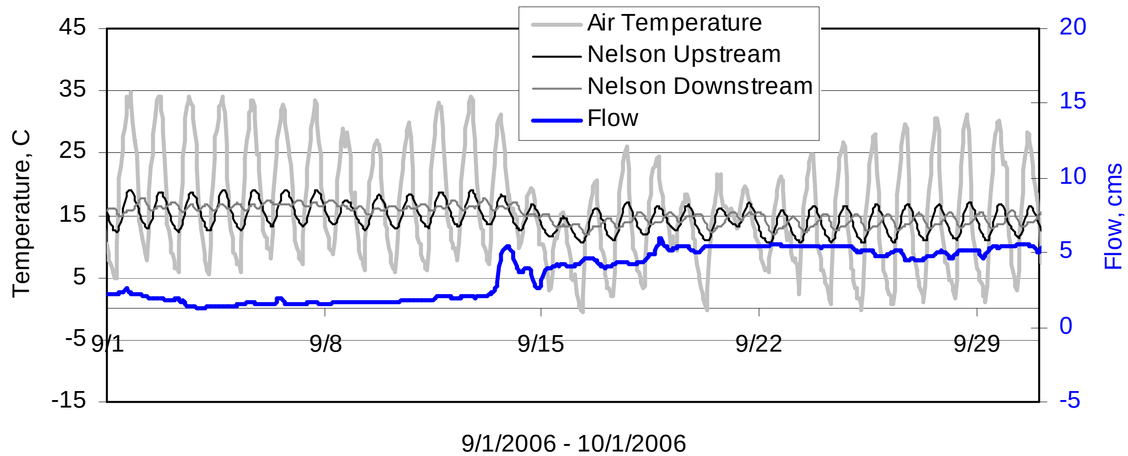


Figure 140. Water temperature, air temperature, and instream flow in the Shasta River; Sept. 2006

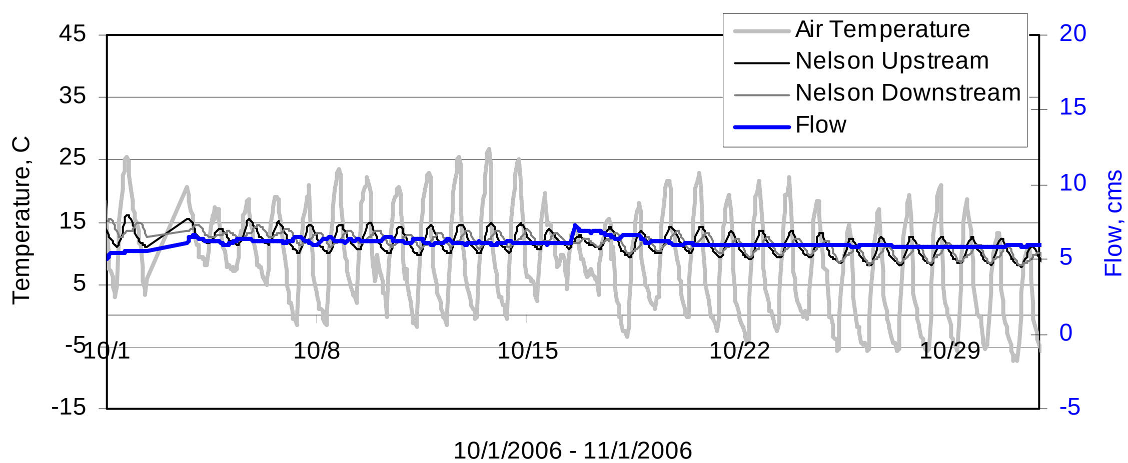


Figure 141. Water temperature, air temperature, and instream flow in the Shasta River; Oct. 2006

170

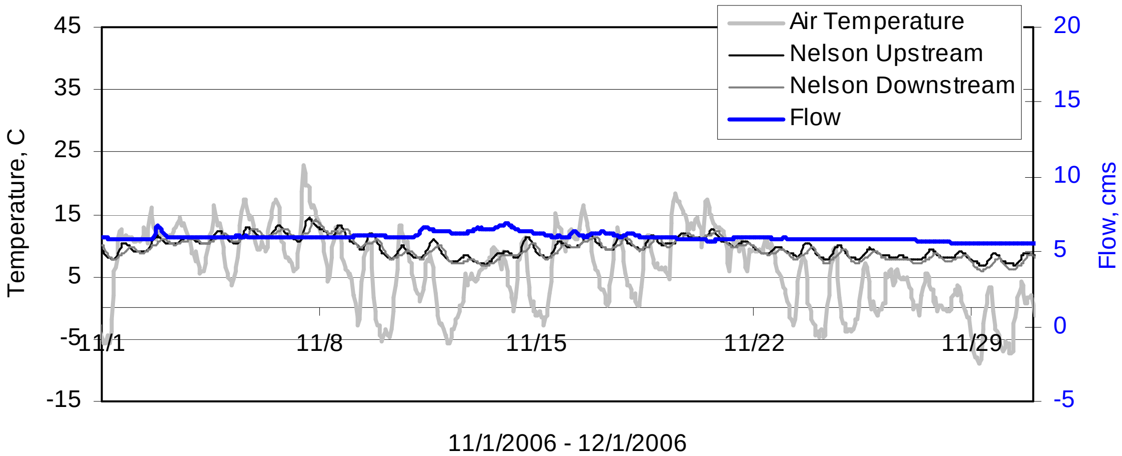


Figure 142. Water temperature, air temperature, and instream flow in the Shasta River; Nov. 2006

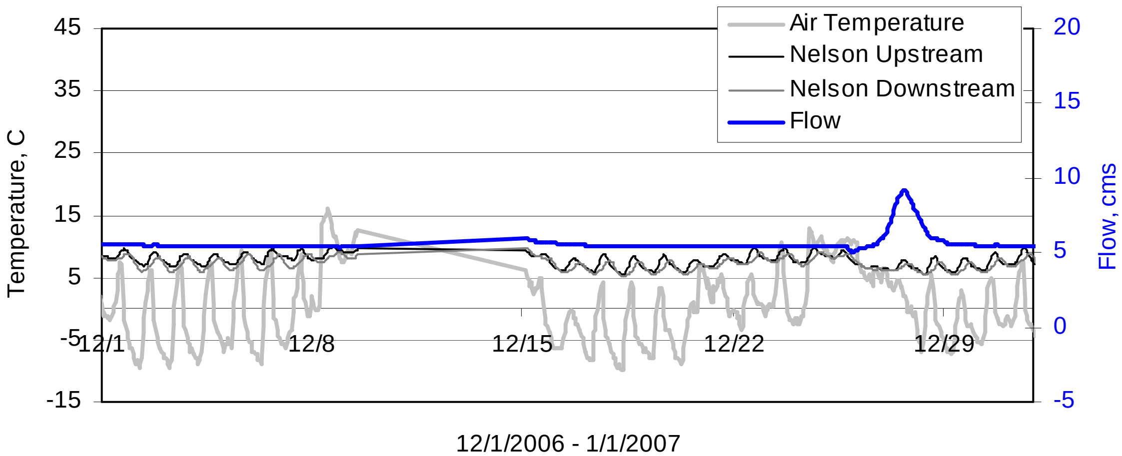


Figure 143. Water temperature, air temperature, and instream flow in the Shasta River; Dec. 2006

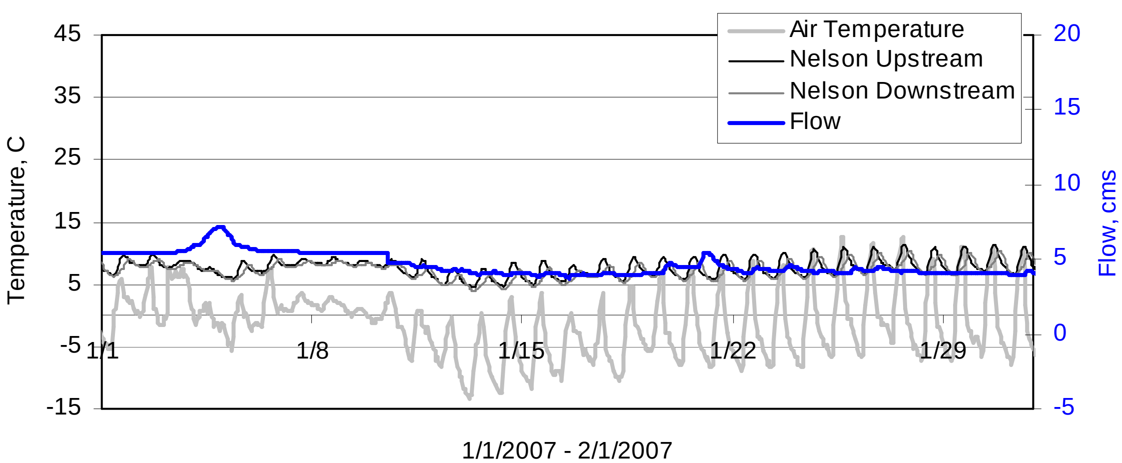


Figure 144. Water temperature, air temperature, and instream flow in the Shasta River; Jan. 2007

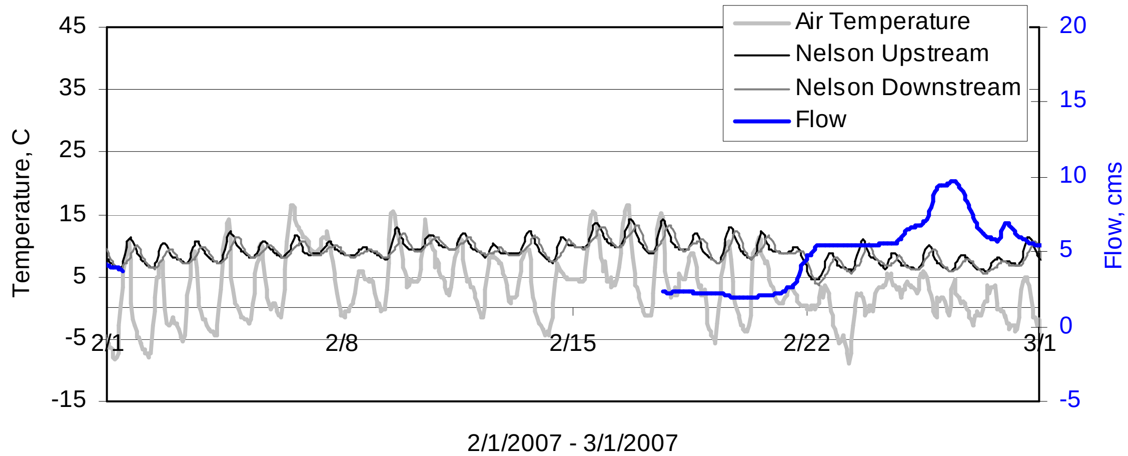


Figure 145. Water temperature, air temperature, and instream flow in the Shasta River; Feb. 2007

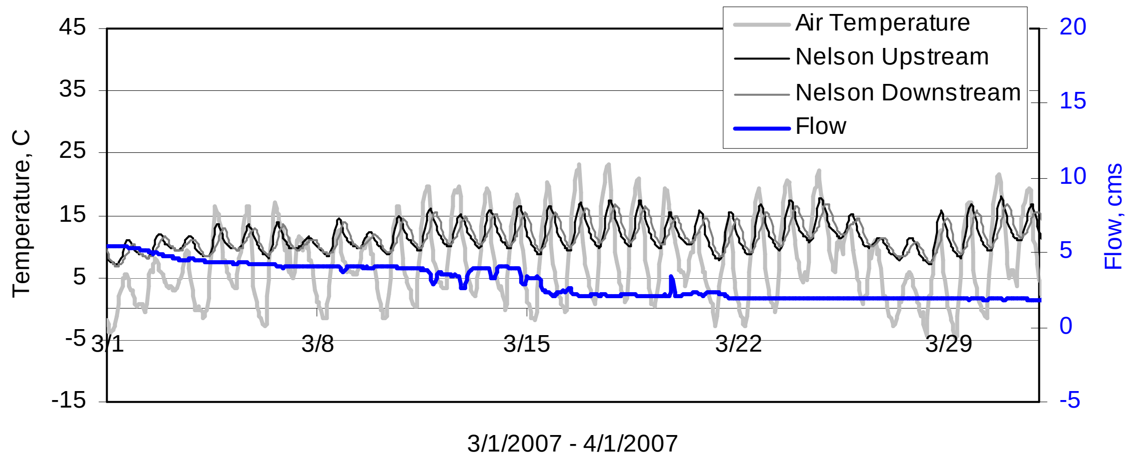


Figure 146. Water temperature, air temperature, and instream flow in the Shasta River; March 2007

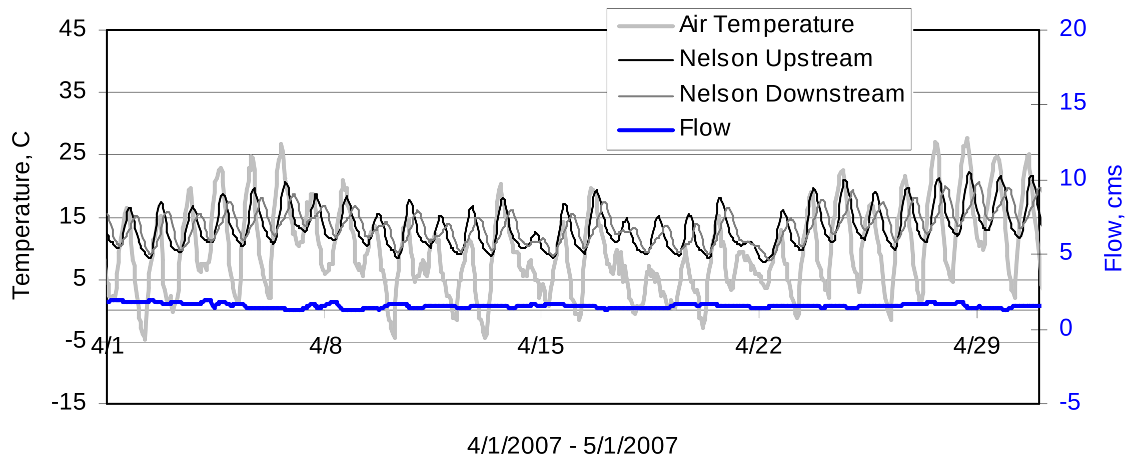


Figure 147. Water temperature, air temperature, and instream flow in the Shasta River; April 2007

172

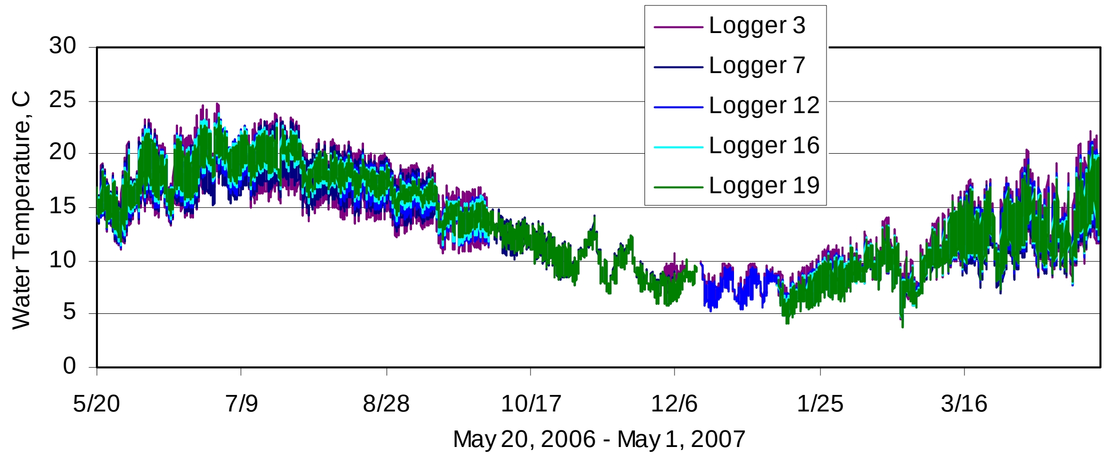


Figure 148. Hourly timeseries of select loggers at Nelson Ranch

Thermal Diversity Exploratory Monitoring – Site Descriptions

Site 1: Spring

Site 1 is a small spring that joins the Shasta River at the upstream end of Nelson Ranch; it was sampled at 10:00 am. The spring emerges from a cut bank approximately 30 m upslope from the Shasta River at 14.7°C. Spring water flows overland toward the river in a channel that is primarily vegetated by grasses and other low vegetation. Cattails grow in the channel where the spring joins the Shasta River. Temperature readings at site 1 were mostly in the 14-16°C range in the longitudinal profile of the spring channel (Figure 149 & Figure 150), and were 14-18°C in the slow moving water around the confluence with the Shasta River (Figure 151). Water depth is included in figures since water temperature in shallow areas is affected by air temperature more quickly than in deeper areas. Bed temperature tended to be slightly more than water temperature, perhaps due to early sample time if the bed was responding more slowly than the water column to changing air temperature and incoming solar radiation.

Note that water temperature scales of sample locations on aerial photos differ because sites were sampled at different times of day. Using a single scale for all sites would reflect temperature changes from diurnal heating rather than the thermal diversity of the Shasta River from seeps, springs, and other inflows.

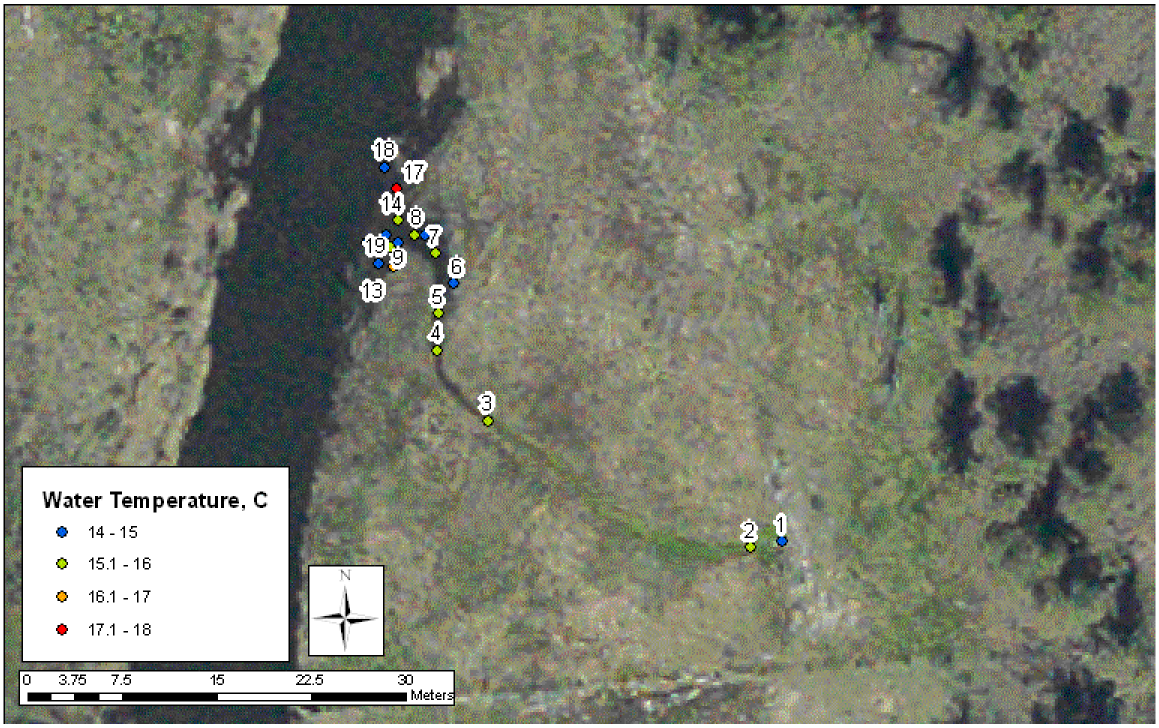


Figure 149. Site 1 sampling locations and temperatures, sampled 8/22/06 10:00

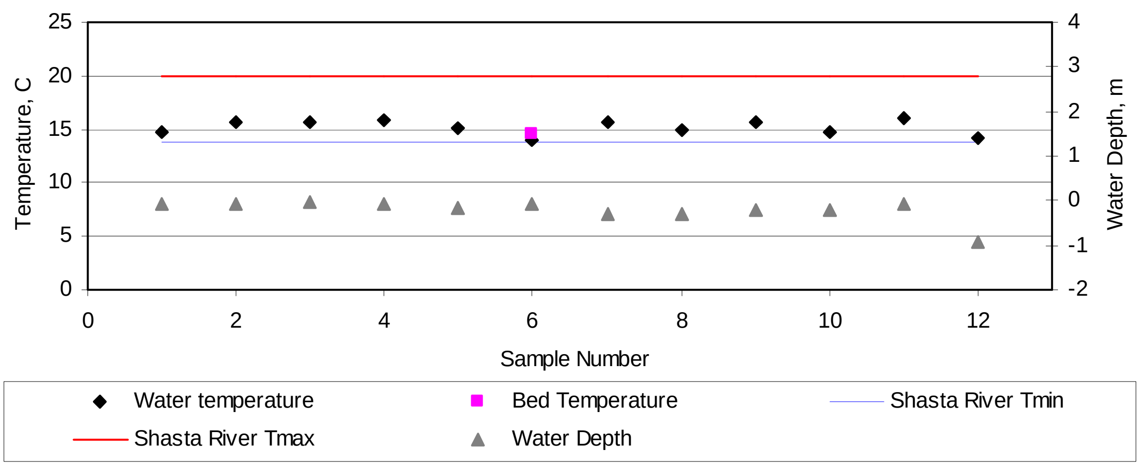


Figure 150. Site 1 longitudinal water temperature sample locations 1-12 (sample 1 taken at spring source, 12 at confluence with Shasta River), sampled 8/22/06 10:00

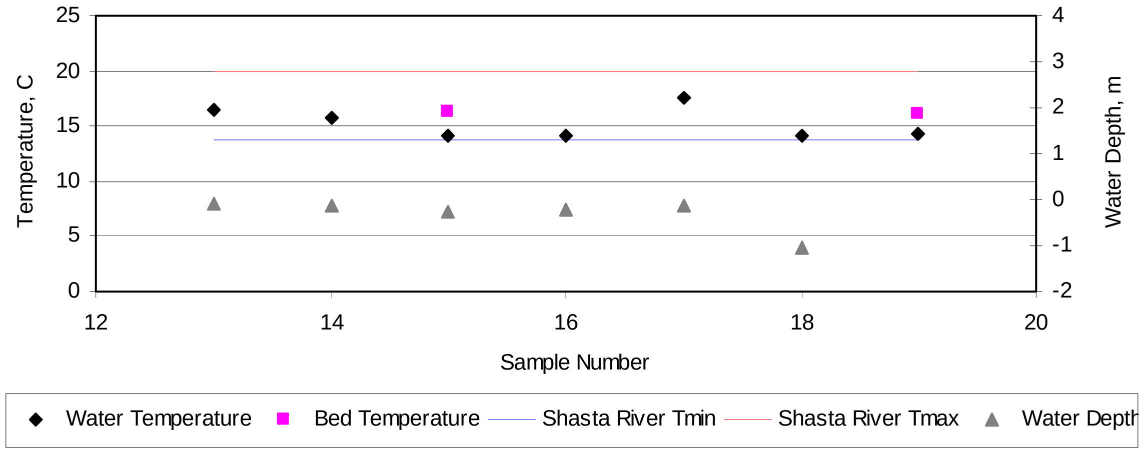


Figure 151. Site 1 thermal variability around confluence with Shasta River sample locations 13-19 (sample 18 taken in the Shasta river), sampled 8/22/06 10:00

Site 2: Pool

Site 2 is a pool adjacent to a cut-bank on river left below a rock diversion dam. This site was sampled at 2:40 pm. Water temperature was sampled with the handheld Acorn unit in a shallow backwater across from site 1 (sample point 8). Here water and bed temperature ranged from 16.4-16.9°C. Woody riparian vegetation between sample point 8 and the Site 2 pool, indicate water may flow subsurface between site 1 and site 2, and not be exposed to atmospheric heating. At site 2, the bank drops steeply to the river, and there is a submerged shelf around the perimeter of the pool that is about a half meter deep. Beyond the submerged shelf, the pool is considerably deeper (estimated at approximately 2 m). Water temperature was sampled mid-column off the submerged shelf at approximately 1 m. Based on the two temperature readings taken below the shelf, water temperature may be slightly cooler below the shelf (up to 1°C). This site likely acts as a small thermal refuge in some conditions. Further exploration is needed to assess whether water flows subsurface from sample point 8 to the pool and to better quantify the size and temperature gradient of the site. With one exception, all temperature readings taken in the pool were between 17-18°C (Figure 152 & Figure 153).

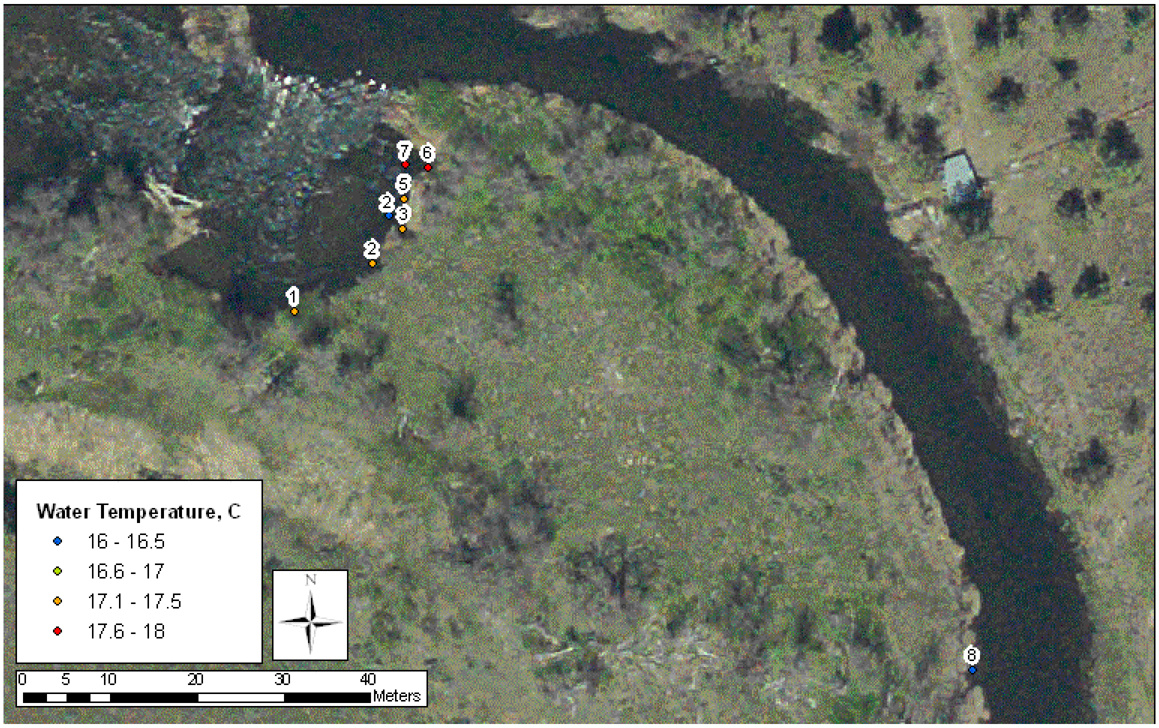


Figure 152. Site 2 sampling locations and temperatures, sampled 8/22/06 14:40

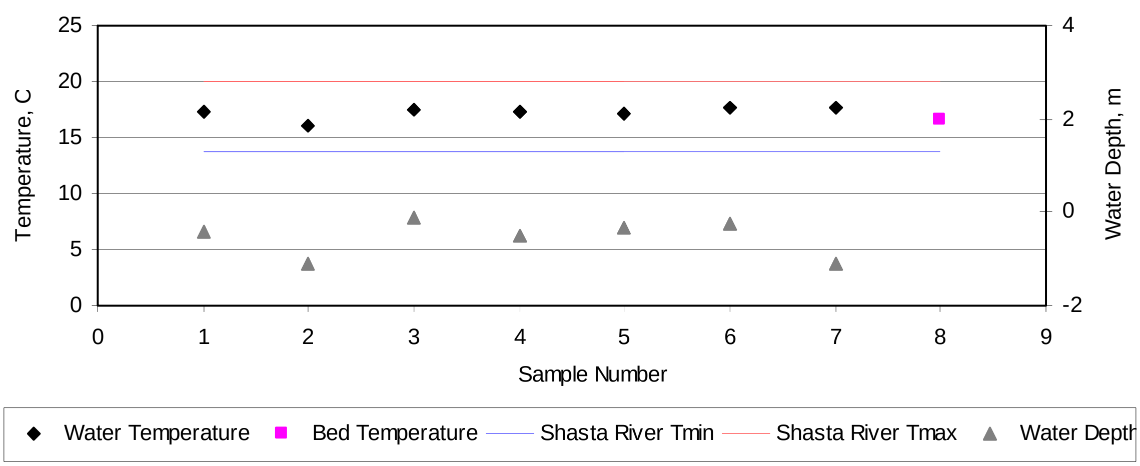


Figure 153. Site 2 water and bed temperature, sampled 8/22/06 14:40

Site 3: Return Flow Ditch

Site 3 is the confluence of the Shasta River and the Nelson Ranch return flow ditch. Almost no vegetation grows in the channel where tailwater joins the river; banks are vegetated by grasses and reeds. Water temperature was measured in the lower portion of the tailwater return channel, the most upstream measurement was taken approximately 15 m from the confluence with the river (Figure 154 & Figure 155). At noon, water temperature in the return ditch varied between 19.2-20.5°C, substantially higher than the 15.2°C Shasta River water, and there was little local thermal variability in the tailwater

(Figure 156). The return flow ditch is shallow, average depth sampled here was 17.8 cm. No bed temperature measurements were taken since water in the shallow ditch can be expect to heat more quickly than the Shasta River, making cool daily subsurface flow unlikely. Below the confluence with the Shasta River, tailwater hugs the bank on river right for approximately 23m downstream. This mixing zone is easily observed by standing in the river below the return ditch or stirring up sediment in the return flow channel.

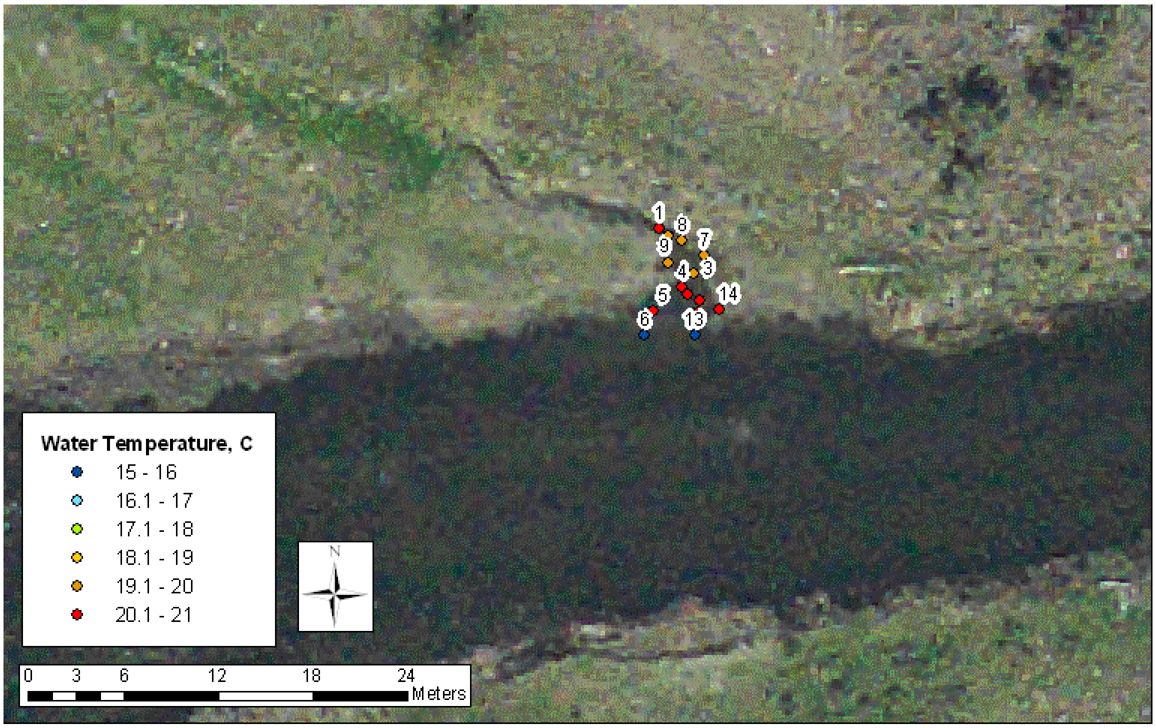


Figure 154. Site 3 sampling locations and temperatures, sampled 8/22/06 12:00

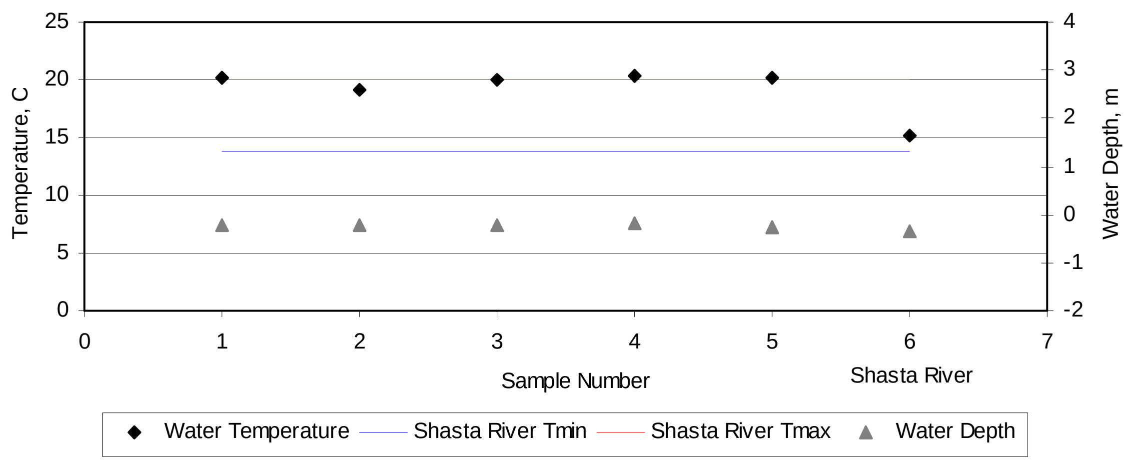


Figure 155. Site 3 longitudinal water temperature (sample 1 taken in tailwater channel, 6 at confluence with Shasta River), sampled 8/22/06 12:00

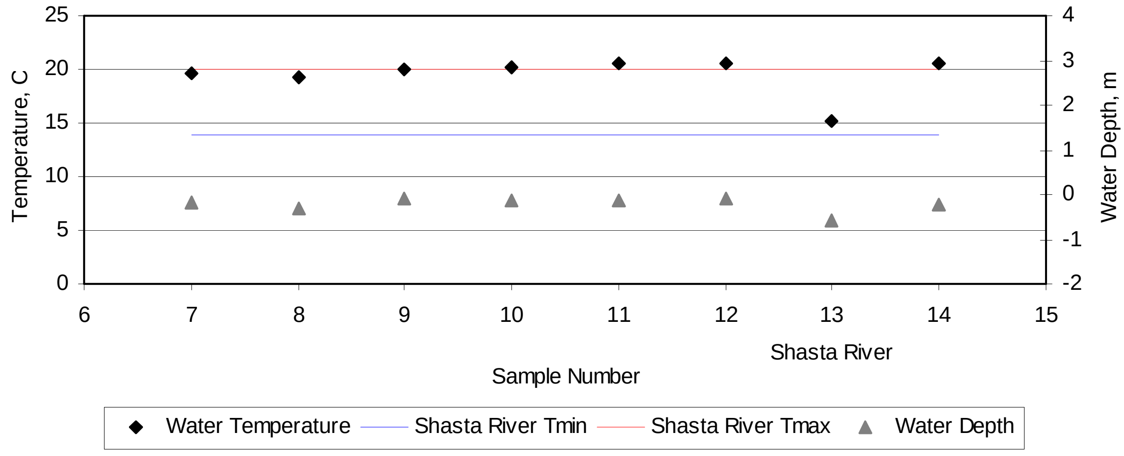


Figure 156. Site 3 thermal variability in the tailwater return channel, sampled 8/22/06 12:00

Site 4: Below screwtrap

This site is a seep or subsurface flow below the CDFG screwtrap and near an old bridge abutment. It was sampled at 3:50 pm. No water visibly runs into this backwater, in fact the channel is dry (soil is moist) within 3m of the Shasta River. This area appears to have little flow, with duckweed, grasses, and reeds growing in shallow water. Water temperature was variable at this site, due in part to the shallow depth and low flow conditions. Regardless, water and bed temperatures of 16-17°C were found, while river temperature was over 18°C (Figure 157 & Figure 158).

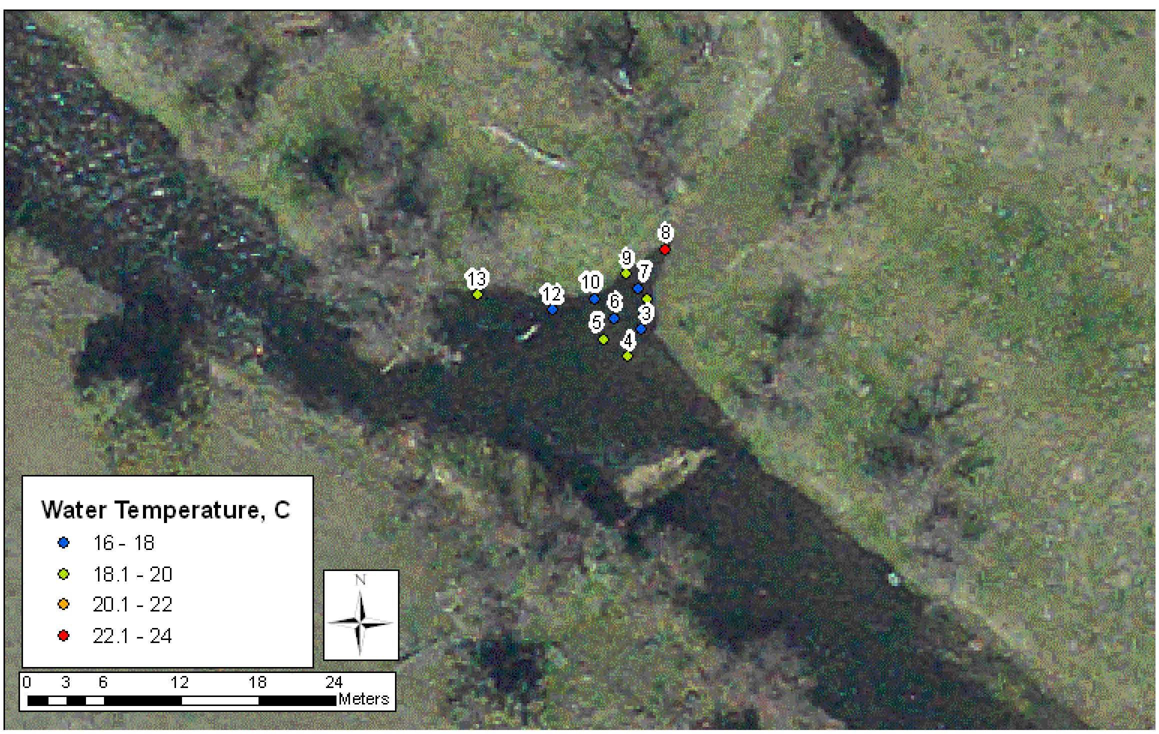


Figure 157. Site 4 sampling locations and temperatures, sampled 8/22/06 15:50

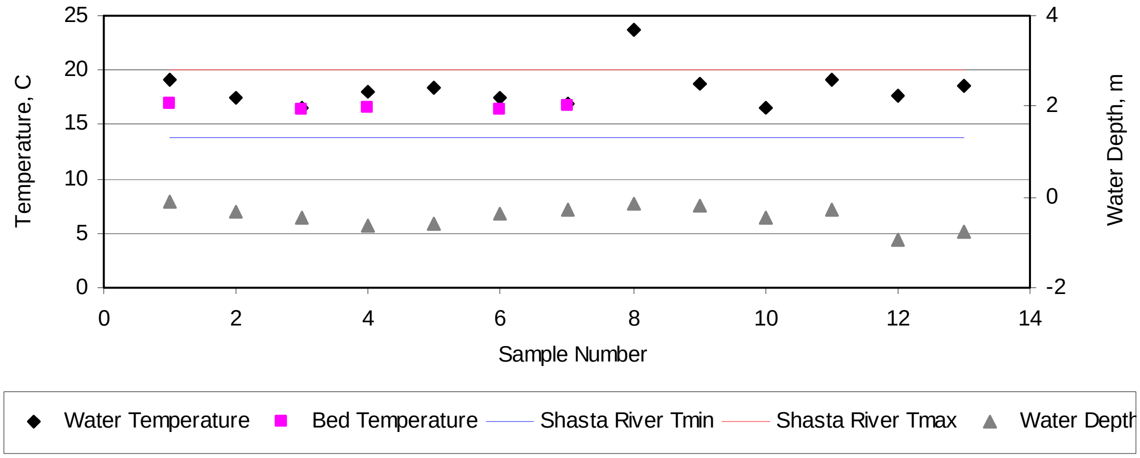


Figure 158. Site 4 water and bed temperature, sampled 8/22/06 15:50

Site 5: Old Channel

A side channel splits from the Shasta River at site 5, and another side channel once joined the Shasta River here. The old channel is now dry but has large riparian trees and herbaceous vegetation. Temperature probing was mainly conducted in river right of the new side channel where subsurface flow from the old side channel may occur. Here water temperature varied between 16-18°C (Figure 159 & Figure 160), with the coolest temperatures in the side channel closest to the mainstem Shasta River. Water temperature increased farther downstream in the side channel. The temperature of the Shasta River was 19°C, although bed temperature was as low as 16.1°C.

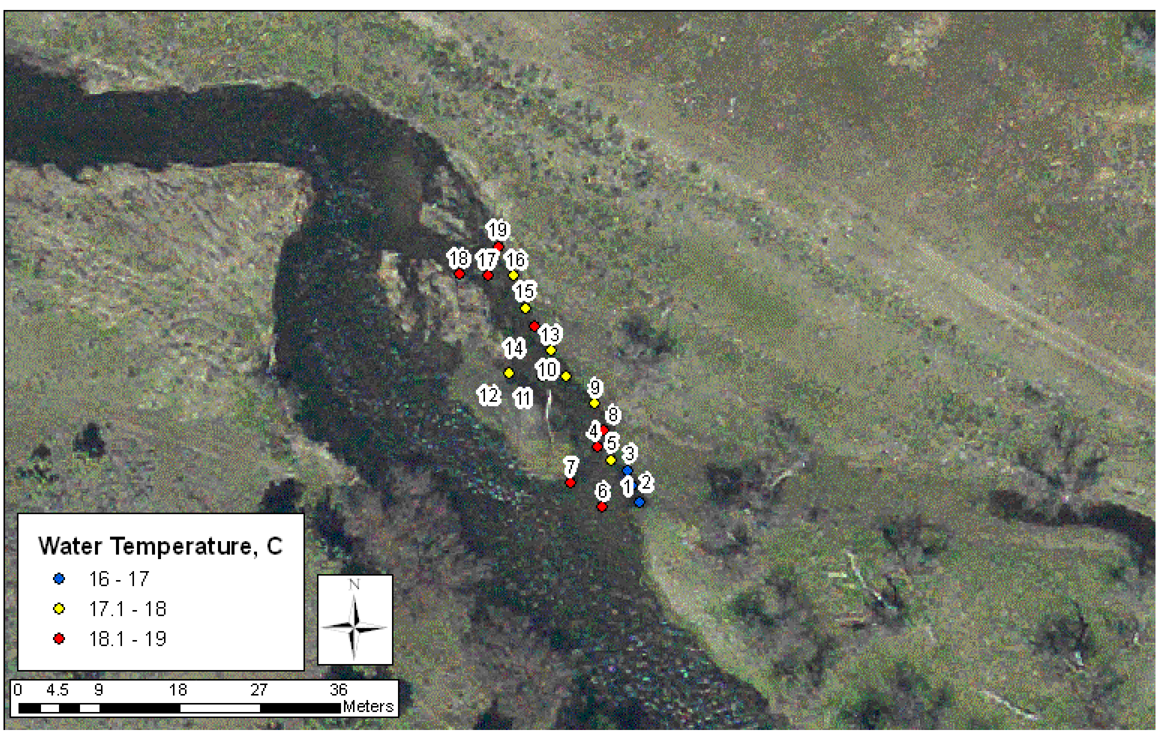


Figure 159. Site 5 sampling locations and temperatures, sampled 8/22/06 16:30

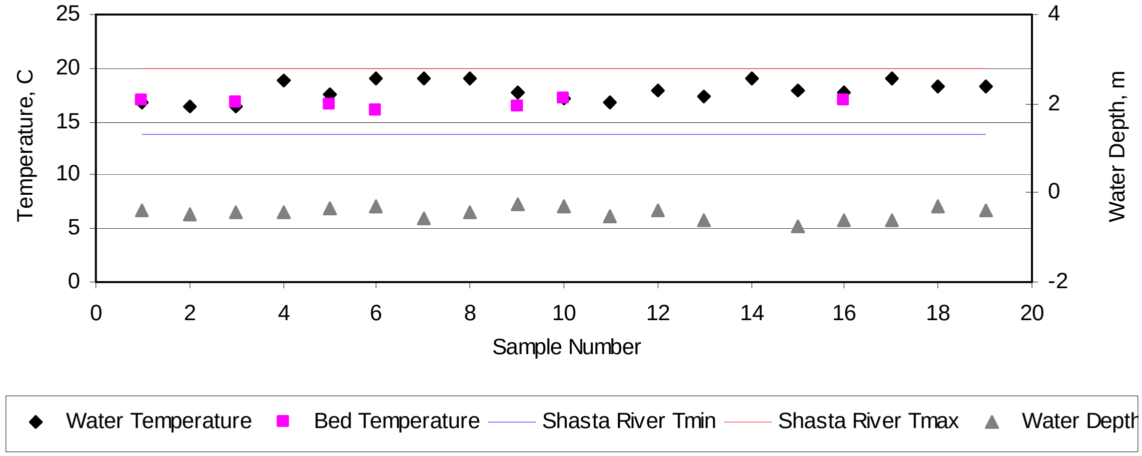


Figure 160. Site 5 water and bed temperature, sampled 8/22/06 16:30

Site 6: Shasta River adjacent to Dream Spring

Site 6 is river right of the Shasta River, down slope of the Dream Spring channel. The site was sampled to locate potential subsurface flow from Dream Springs at 2:00 pm. This site has high reeds and cattails along much of the Shasta River, making access difficult. Beyond the reeds and cattails, the Shasta River is deep (> 1.5m). In front of the reeds are shallow backwater areas that generally have little flow and are vegetated with duckweed. Overall, it was difficult to determine whether backwaters were supplied via subsurface flow, or whether they were derived from the Shasta River. However, the water in the backwaters was warm, ranging from 17-19°C (Figure 161 & Figure 162), suggesting limited subsurface inflow. The temperature of the Shasta River was 16-17°C, and was sampled mid-water column. This site did not show obvious signs or warm or cool inflows, rather water temperature is probably primarily influenced by atmospheric conditions.

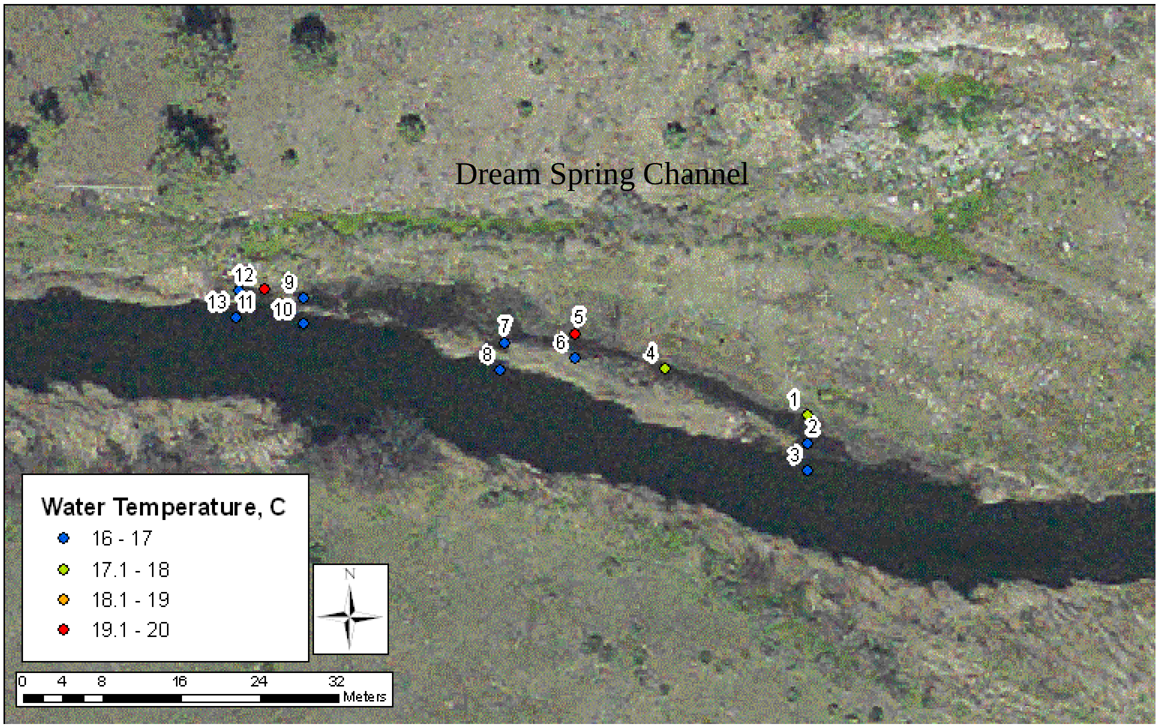


Figure 161. Site 6 sampling locations and temperatures, sampled 8/23/06 14:00

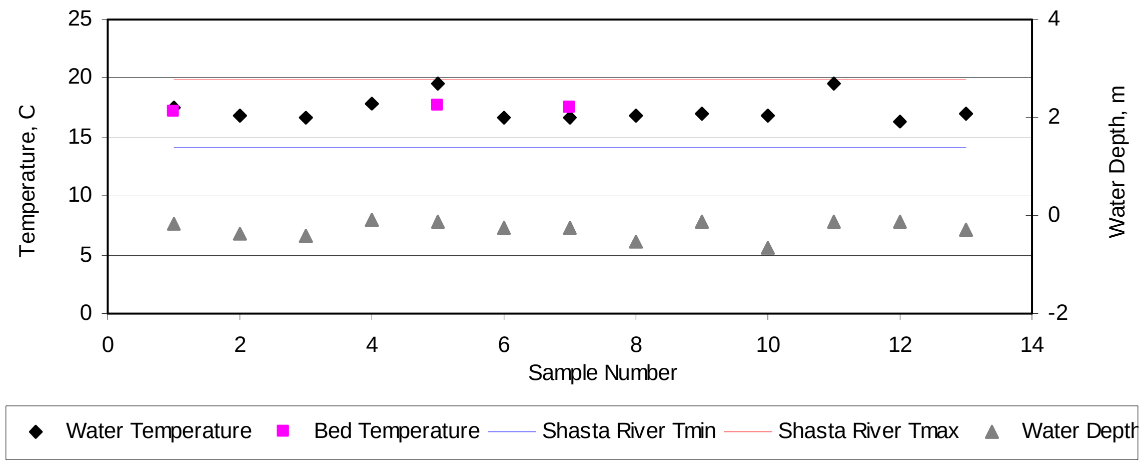


Figure 162. Site 6 water and bed temperature, sampled 8/23/06 14:00

Site 7: Dream Spring Longitudinal Profile

At 1:00 pm, Dream Spring was 16.3°C at its source approximately 30 m directly uphill of the Shasta River, and was 19.6°C where it joined the Shasta River. Dream Spring flows into an unused irrigation ditch parallel to the Shasta River for approximately 100 m before flowing into the Shasta River. The average depth of Dream Spring in the irrigation channel is less than 15 cm. Nettles, greasewood, or emergent aquatic vegetation shades sections of the channel, and sections of the channel are open to sunlight. This profile shows water from Dream Springs heating considerably between its source and confluence with the Shasta River (Figure 163 & Figure 164). At 1:10 pm, the time of sampling,

181

Shasta River water was 17°C, cooler than Dream Springs at its confluence with the Shasta River. While beginning cooler than the mainstem on this day, this spring heated to more than the mainstem temperature on it's overland route to the mainstem.

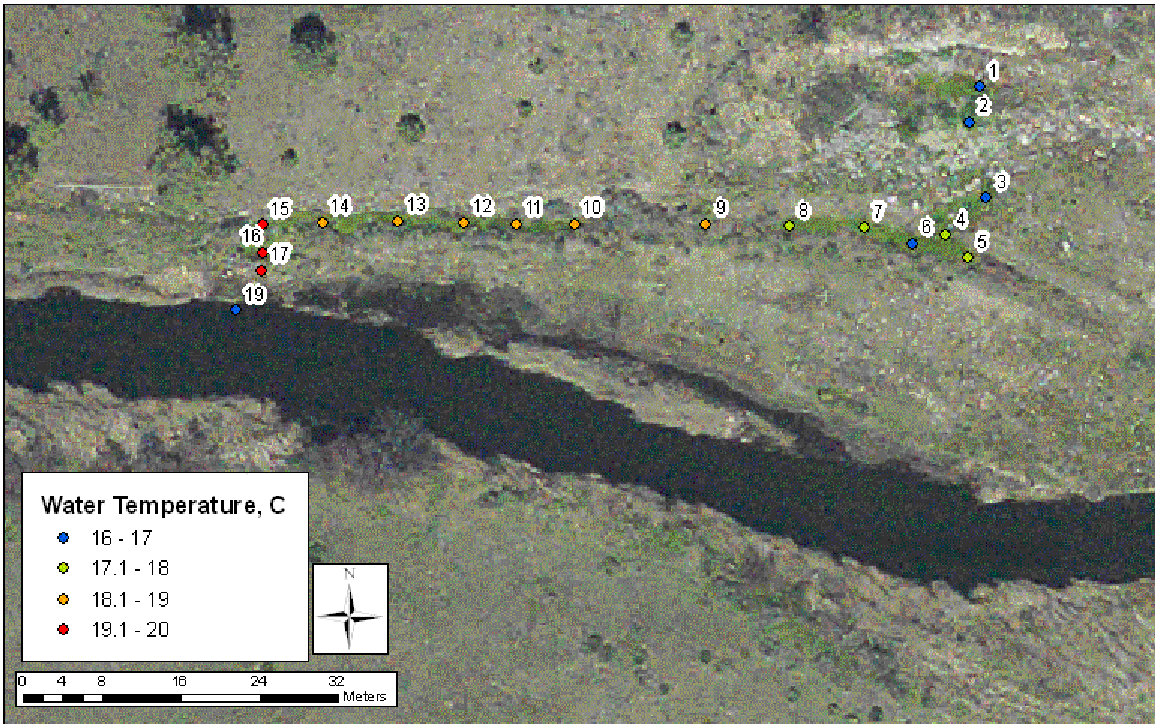


Figure 163. Dream Spring sampling locations and temperatures, sampled 8/23/06 13:00

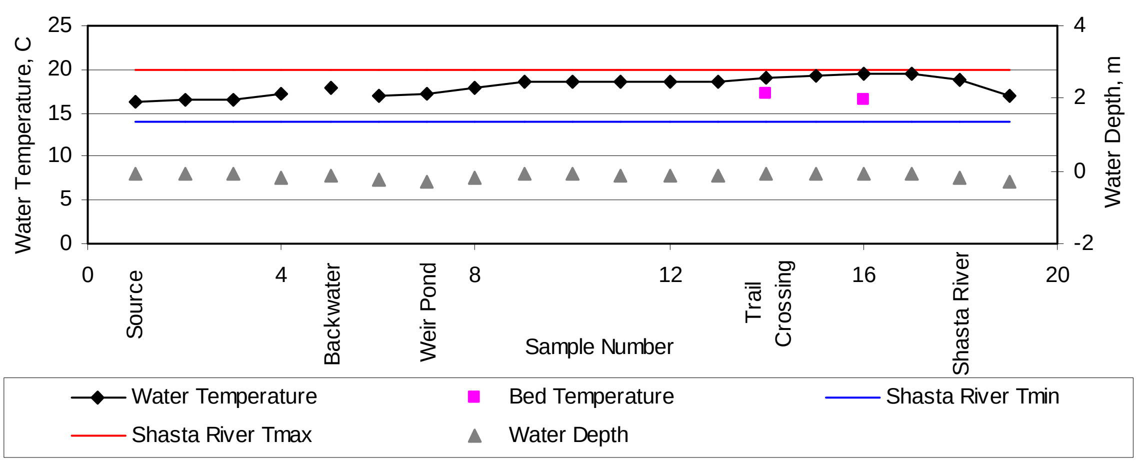


Figure 164. Dream spring water temperature, bed temperature, and depth, sampled 8/23/06 13:00

Winter and Spring Lateral Transects

Transect 1 illustration and upper and lower maximum temperature chart in main text, (Chapter 3).

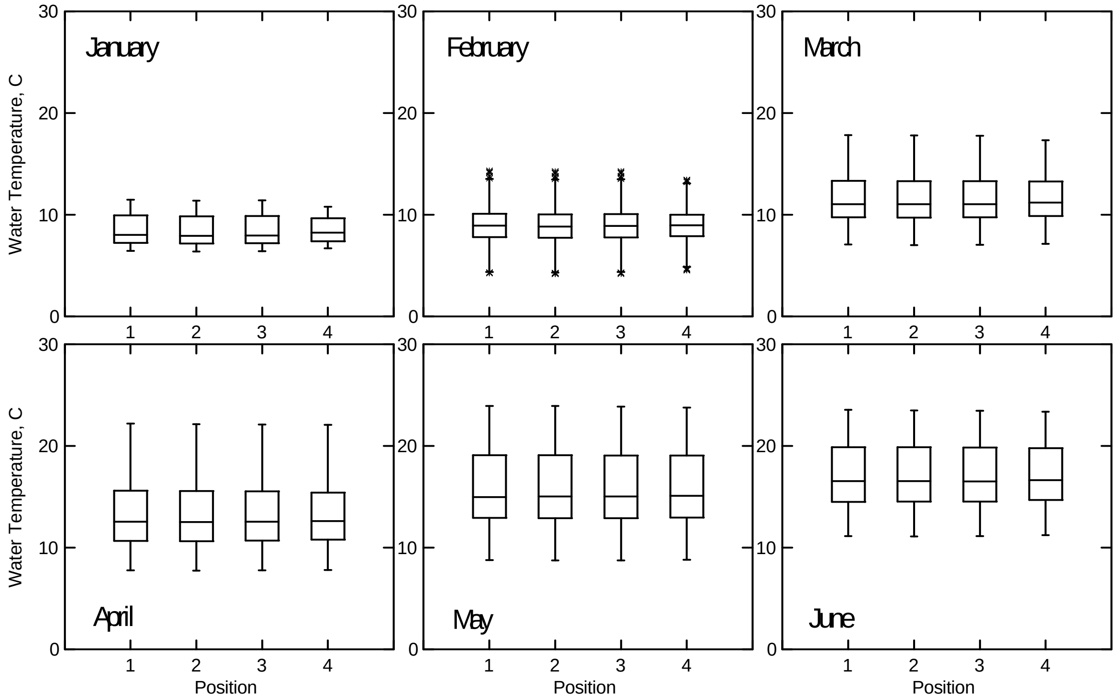


Figure 165. Transect 1 water temperature by logger position and month

Transect 2

Transect 2 was located downstream of the GID diversion structure, at a shallow bench with dead reeds and cattail, bordered by willows. From the bench, the main channel drops off steeply. There is a large pool on the opposite side of the river. Three loggers were placed on the shallow bench, another logger was placed in the bank of the main channel (Figure 166, Table 24). The last logger was connected to a 3m cable and placed in the main current. No handheld measurement was taken for this logger due to river depth and velocity. Loggers 2-4 and 2-5 were removed on 3/20/07 because the bench was dry in that location. The temperature signal for logger 2-3 could not be used after 4/3/07 because it became exposed to air. Thus, after 4/3 this transect had only two loggers, located in the main channel.

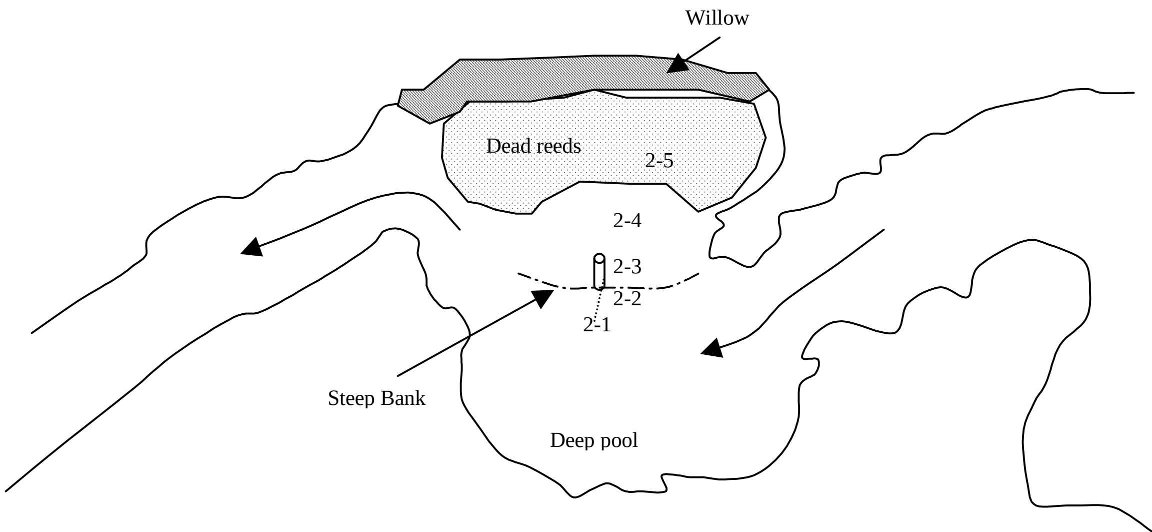


Figure 166. Transect 2 – Schematic of shallow bench below GID diversion

Table 24. Transect 2 logger depth and water temperature at deployment

Logger ID	Depth (m)	Water Temperature (C)
2-4	0.1	7.3
2-3	0.13	7.3
2-5	0.05	8.3
2-2	0.56	7.3

Results from logger 2-4 was removed from the dataset because it was strikingly less variable that the other loggers in the transect, indicating it may have become covered in sediment. Generally, loggers 2-1 and 2-2 were nearly identical, and most temperature differences were from loggers 3 and 5 (Figure 167). There is negligible difference in maximum water temperature after loggers 2-3, 2-4, and 2-5 were removed in mid-April.

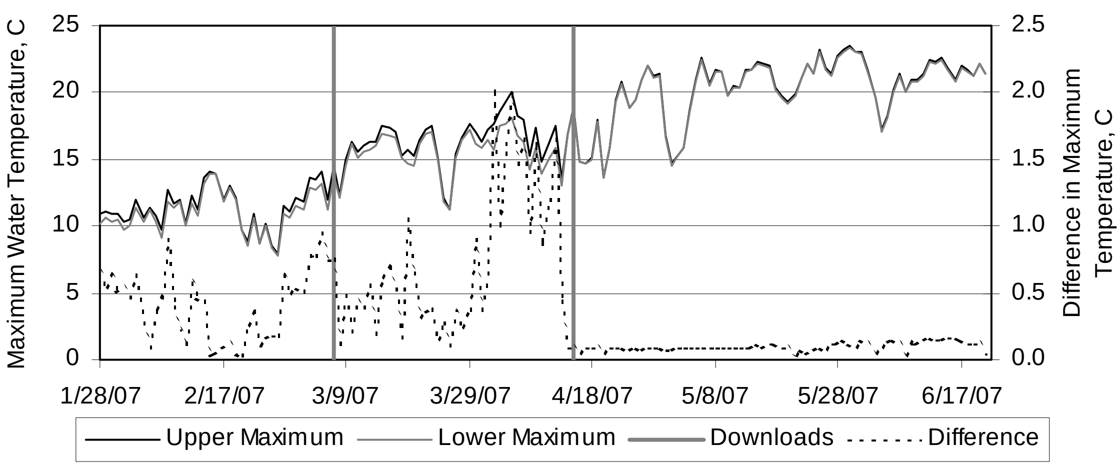


Figure 167. Transect 2 upper, lower, and difference in maximum daily water temperature

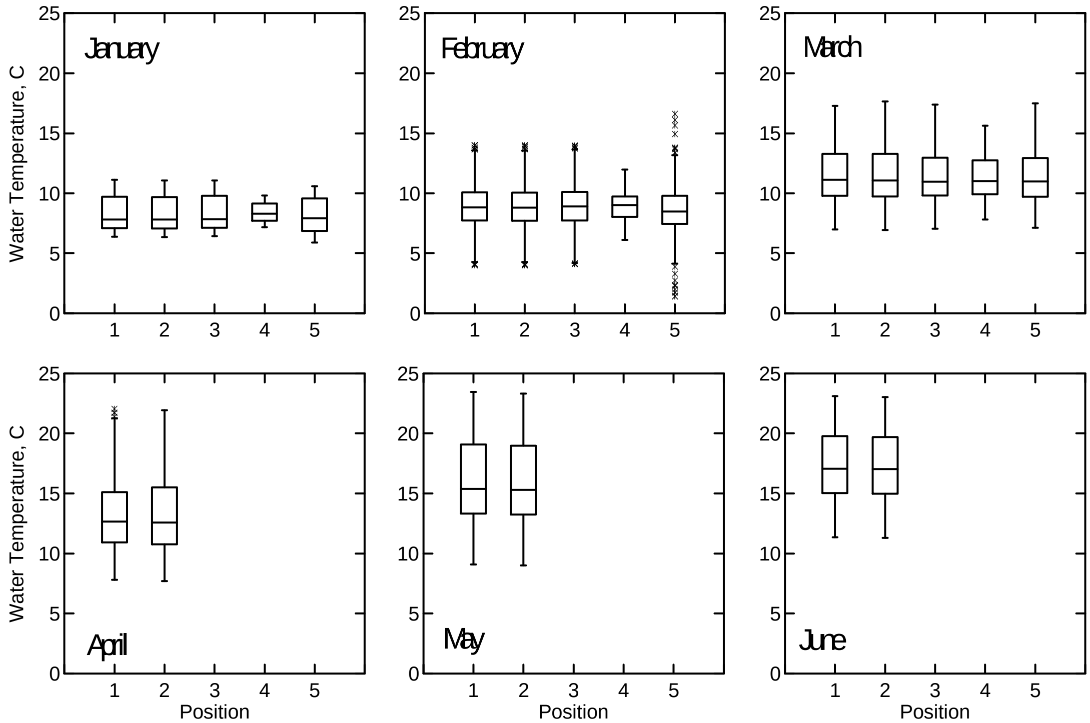


Figure 168. Transect 2 water temperature by logger position and month

Transect 3

Transect 3 was a deep-channel site with a steep bank located approximately 200 yards downstream of transect 2. Chinook redds have been observed at this site (C. Jeffres, pers. comm., 12/2006), indicating possible spawning habitat for coho salmon. Due to the depth, only three loggers were installed at this site (Figure 169). Loggers 3-3 and 3-2 were connected to the bank wall (Table 25), and logger 3-1 was connected to a 3m cable staked to the bank and set in the channel.

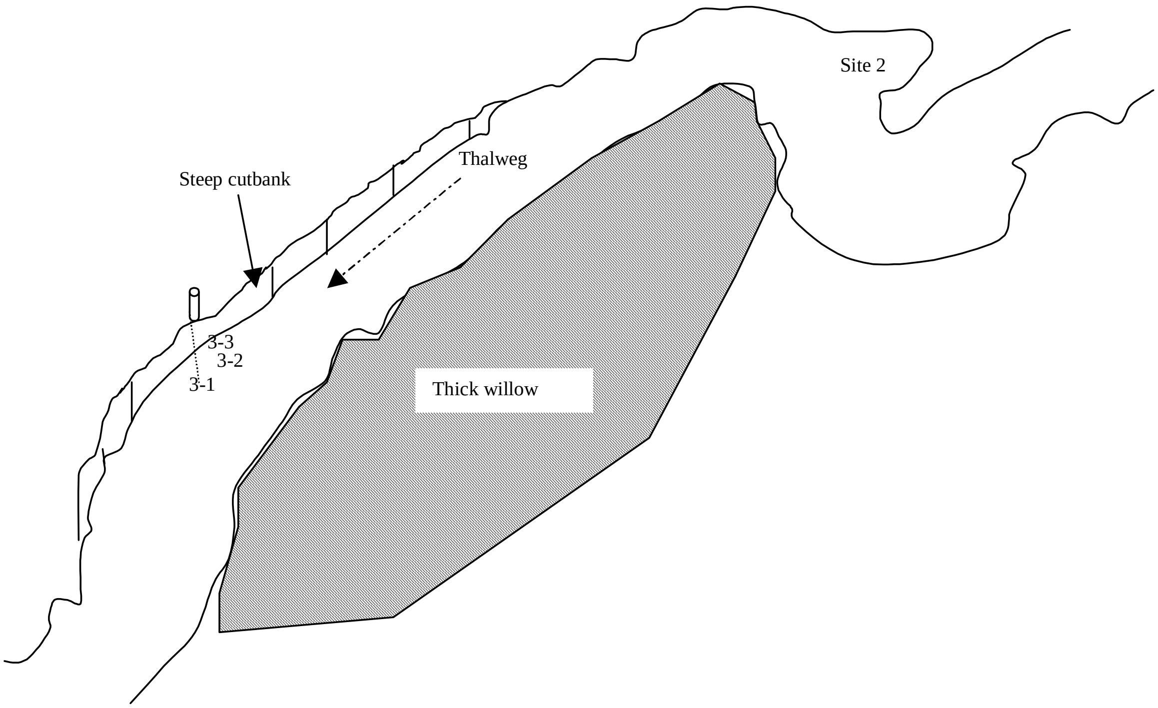


Figure 169. Transect 3 – Schematic of deep channel below GID diversion

Table 25. Transect 3 logger depth and water temperature at deployment

Logger ID	Depth (m)	Water Temperature (C)
3-3	0.46	8.2
3-2	0.71	8.2

Transect 3 shows almost no thermal variability between transect positions (Figure 170). There is a small uniform difference between daily maximum temperature that may have been logger error (it is below the 0.2°C accuracy limit for the thermistors).

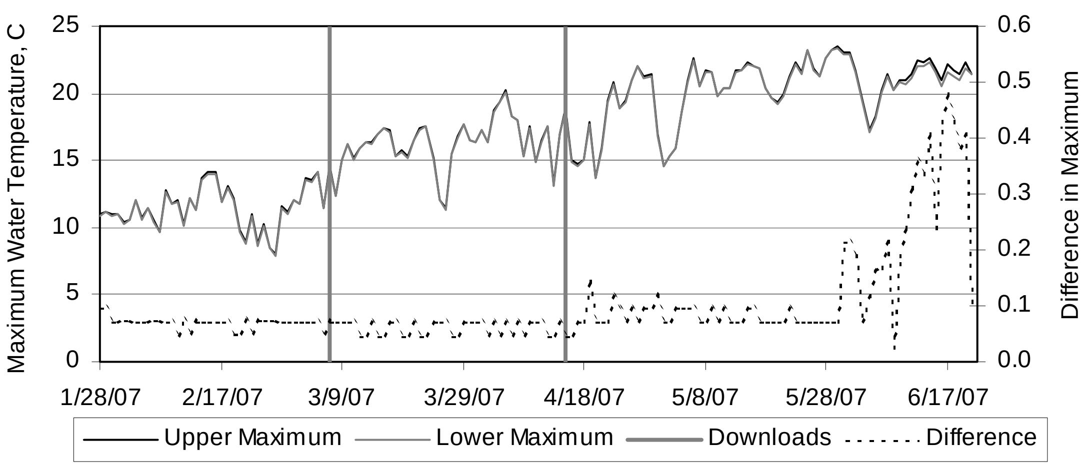


Figure 170. Transect 3 upper, lower, and difference in maximum daily water temperature

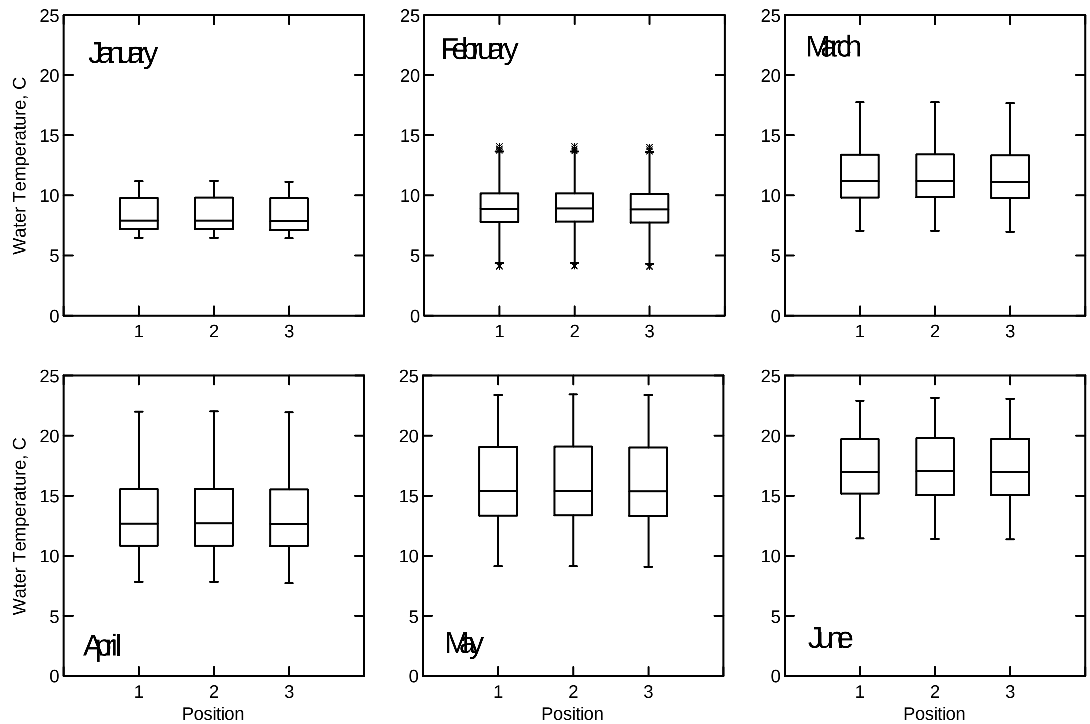


Figure 171. Transect 3 water temperature by logger position and month

Transect 4

Transect 4 was in slow moving water where the channel gradually deepens, creating a margin at river right of relatively shallow habitat. At that site, the thalweg is closer to the left side of the channel (Figure 172). The temperature transect was installed in an area with few shrubs, although shrubs were present immediately upstream and downstream of the site, and herbaceous vegetation was present at the site in spring. Loggers 4-4 and 4-3 were on short cables near the riverbank, and loggers 4-2 and 4-1 were connected to 1.5m and 3m cables respectively and placed in the main channel (Table 26).

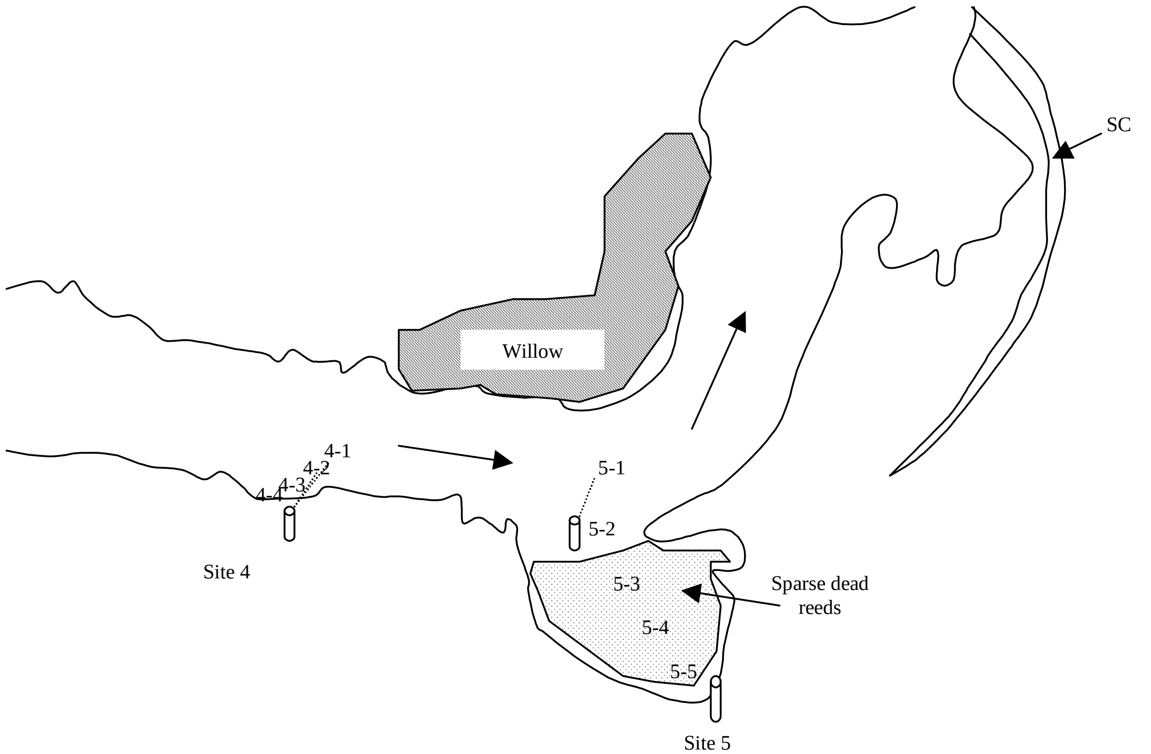


Figure 172. Schematic of transect 4 - deep channel habitat, and transect 5 – shallow bench near side channel 1

Table 26. Transect 4 logger depth and water temperature at deployment

Logger ID	Depth (m)	Water Temperature (C)
4-4	0.13	10.3
4-3	0.36	8.7
4-2	--	10.5 (approximate location)

Results from loggers 4-4 and 4-2 show a muted temperature signal, suggesting these devices were covered in sediment in the low velocity conditions at this site. Average difference in daily maximum water temperature along transect 4 was 3.3°C (Figure 173). However, there was no discernable pattern relating temperature difference to logger position or distance from shore (Figure 174).

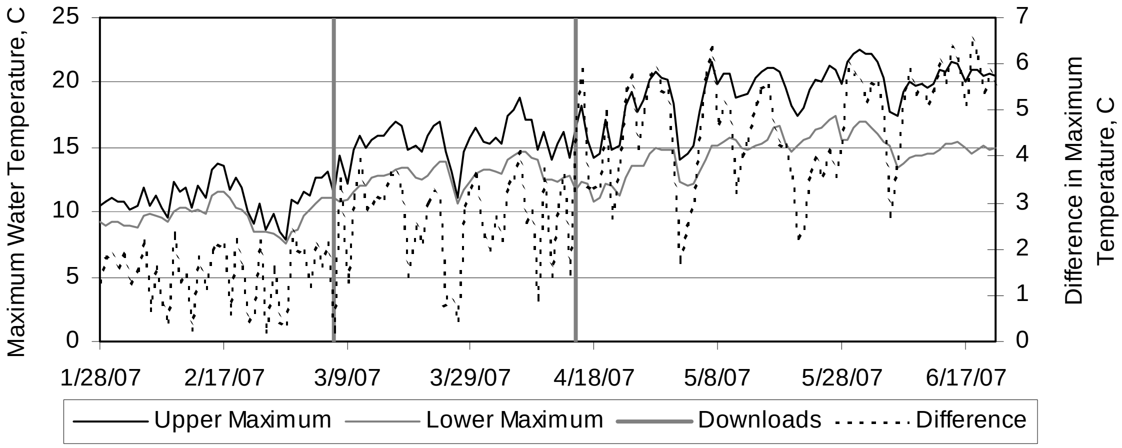


Figure 173. Transect 4 upper, lower, and difference in maximum daily water temperature, and times devices were downloaded and replaced

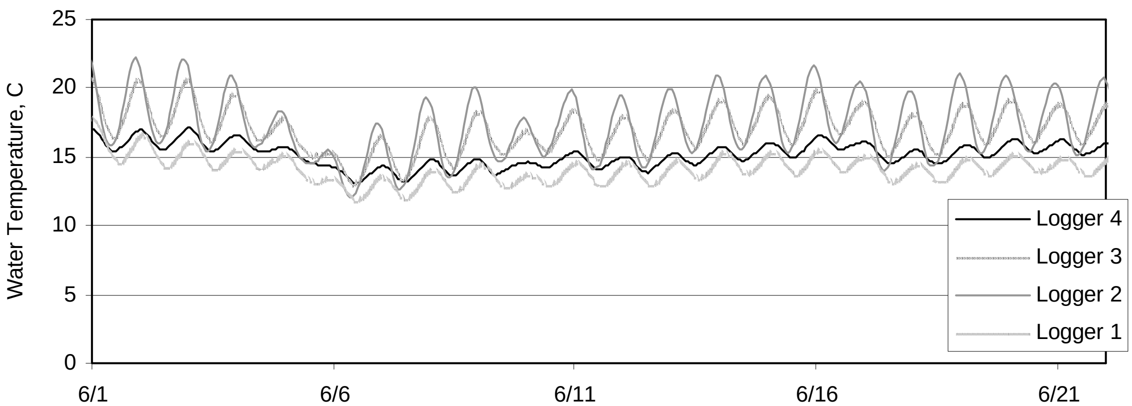


Figure 174. Transect 4 hourly water temperature, June 1 - June 22, 2007

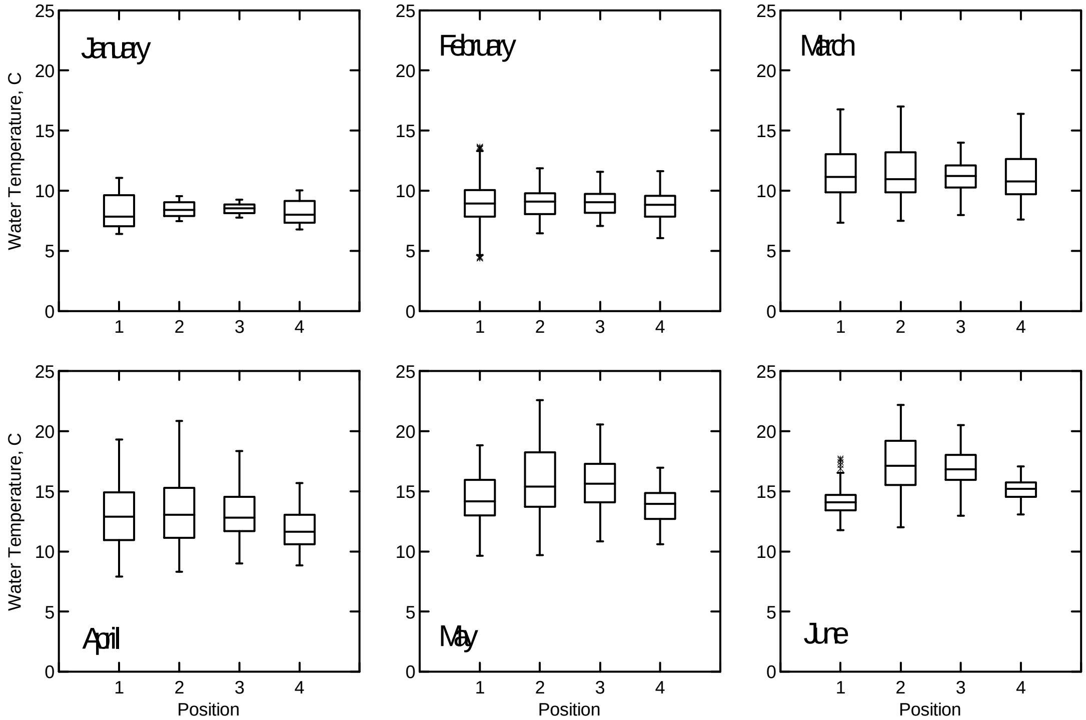


Figure 175. Transect 4 water temperature by logger position and month

Transect 5

Transect 5 was approximately 50m downstream of site 4 near the end of the dirt road leading to GID. This site was a shallow backwater bench with a steep drop-off to the main channel of the river, where the current increased (Figure 172). The backwater appears to be an inlet for a side channel when river stage is high, and may receive subsurface flow under some conditions. Dead vegetation, and some logs and stumps occupied the backwater. Four loggers were deployed in a line across the bench, and the shallow edges of the backwater were frozen at the time of deployment (Table 27). Logger 5-1 was connected to a stake on a 3m cable and placed in the main channel. Loggers placed on the bench farthest from the river (5-3, 5-4, and 5-5) became exposed to air in March, after which only two loggers recorded water temperature in this transect.

Table 27. Transect 5 logger depth and water temperature at deployment

Logger ID	Depth (m)	Water Temperature (C)
5-5	0.16	5.7
5-4	0.16	9.4
5-3	0.08	10.7
5-2	0.28	10.1

Transect 5 had a modest amount of thermal variability when all five loggers were submerged from 1/28 to 3/9, 2007 (Figure 176). Beginning on March 10, the three loggers located on the floodplain became exposed to air and those records were removed. From mid-March to mid-April, the remaining two loggers were nearly identical. Beginning April 17, the two remaining loggers have drastically different temperature patterns and timing (Figure 177). Most likely, logger 2 was placed in very shallow water (especially during low river stage periods), and was more sensitive to air temperature and solar radiation than logger 1.

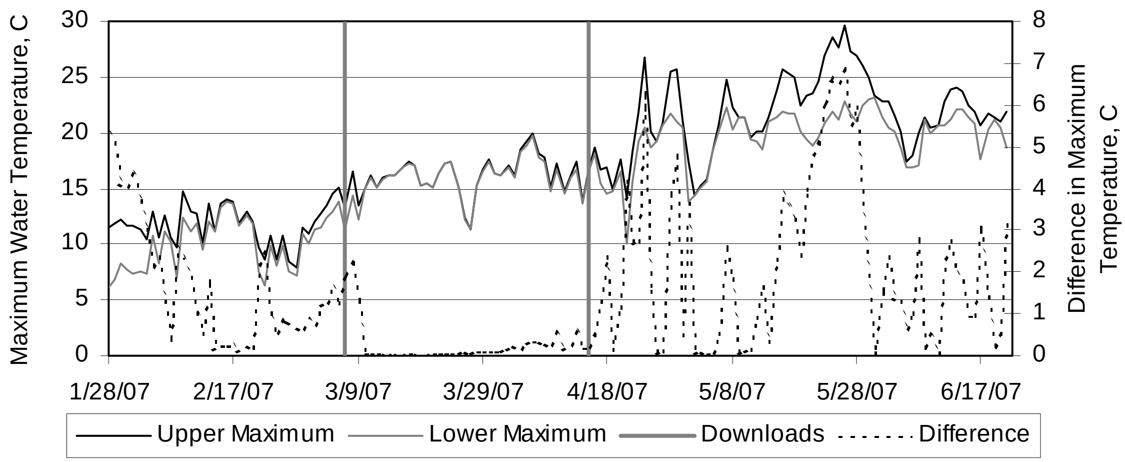


Figure 176. Transect 5 upper, lower, and difference in maximum daily water temperature, and times devices were downloaded and replaced

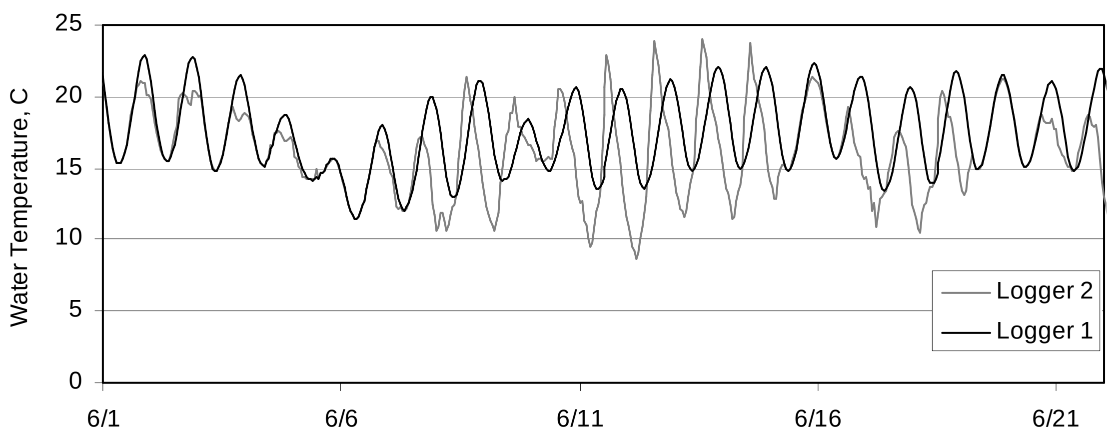


Figure 177. Transect 5 hourly water temperature, June 1 - June 22, 2007

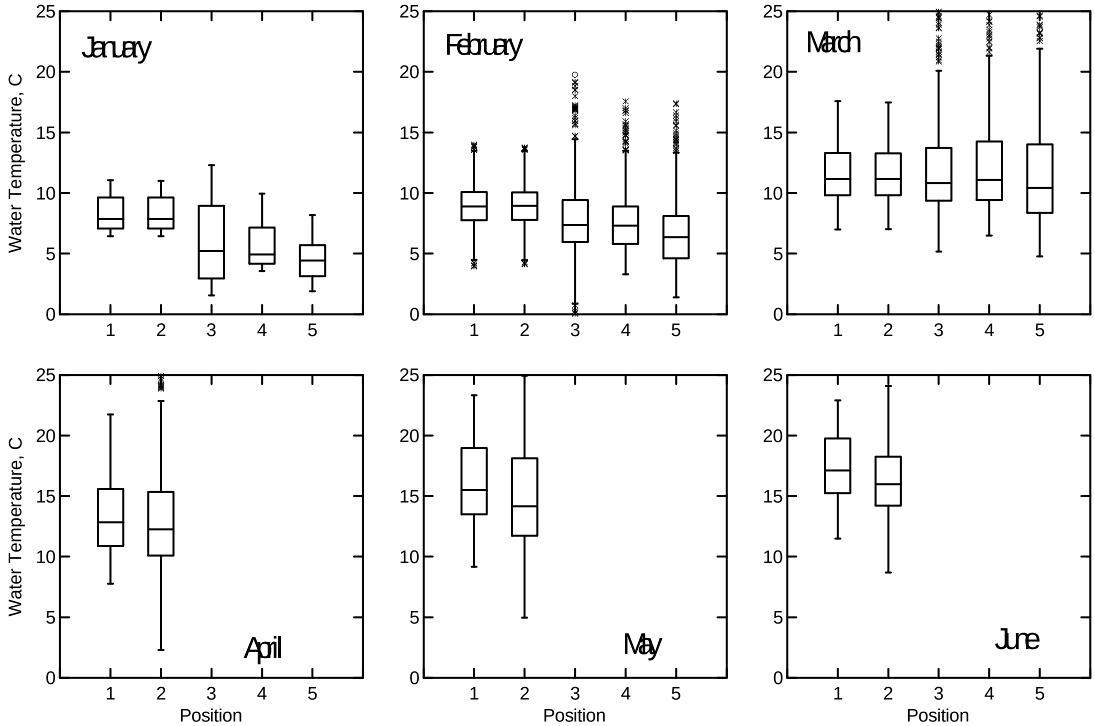


Figure 178. Transect 5 water temperature by logger position and month

Transect 6

Transect 6 is located in a shallow portion of river along the inside curve of a meander bend (it is also near UC Davis' transect #8). The thalweg is near the opposite side of the channel at the outside of the meander. Three loggers were attached to short cables in a line, and a fourth was attached to a long cable near the thalweg (Figure 179 and Table 28).

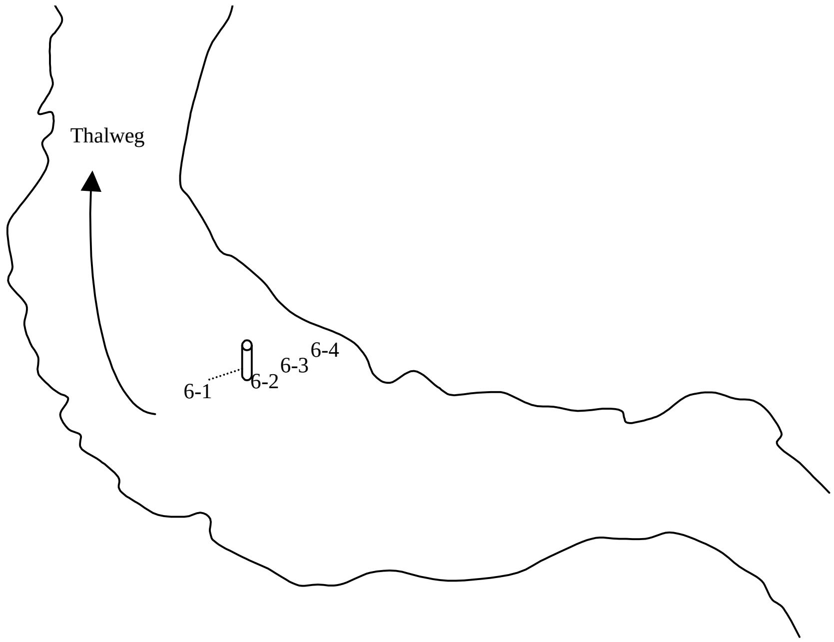


Figure 179. Transect 6 – Schematic of shallow inner channel near UC Davis' cross-section #8

Table 28. Transect 6 logger depth and water temperature at deployment

Logger ID	Depth (m)	Water Temperature (C)
6-2	0.41	8.2
6-3	0.23	8.2
6-4	0.25	8.2

Transect 6 was one of the most interesting transects because thermal variability along the transect increases into the summer months (Figure 180). Logger 2 has less variability than the other loggers and is primarily responsible for this effect (Figure 181). Transect 6, in shallow water habitat, is more likely than deep water transects to have small lateral temperature differences, especially after floodplains and other distinct habitat features have dried for the season. However, thermal diversity occurring in summer that increases maximum water temperature, but that doesn't provide cooler habitat in winter and spring, is likely to be of little use to salmon and may reduce overall habitat quality.

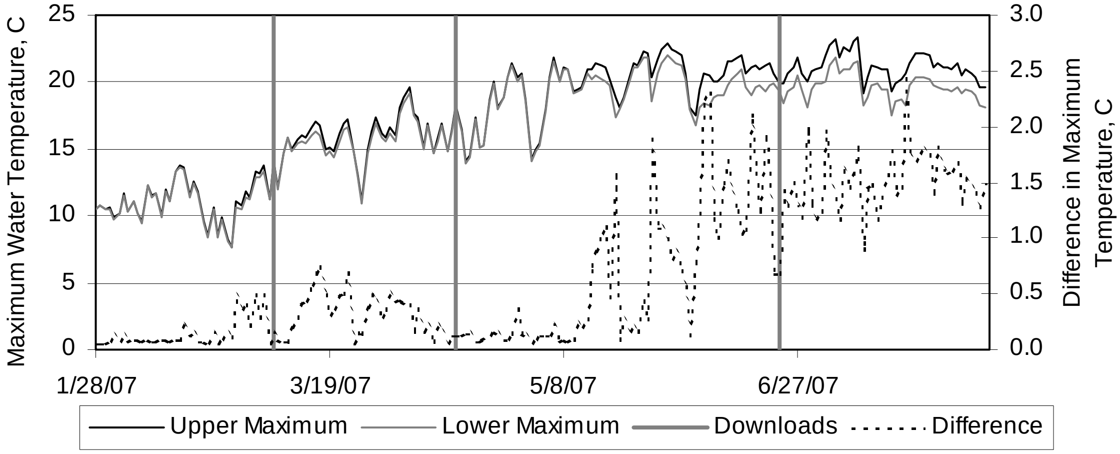


Figure 180. Transect 6 upper, lower, and difference in maximum daily water temperature, and times devices were downloaded and replaced

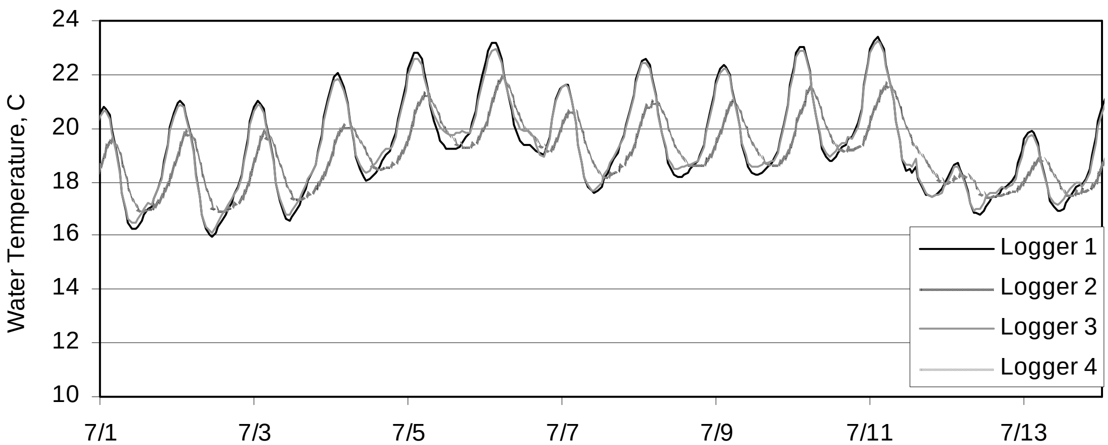


Figure 181. Transect 6 hourly water temperature, July 1 – July 14, 2007

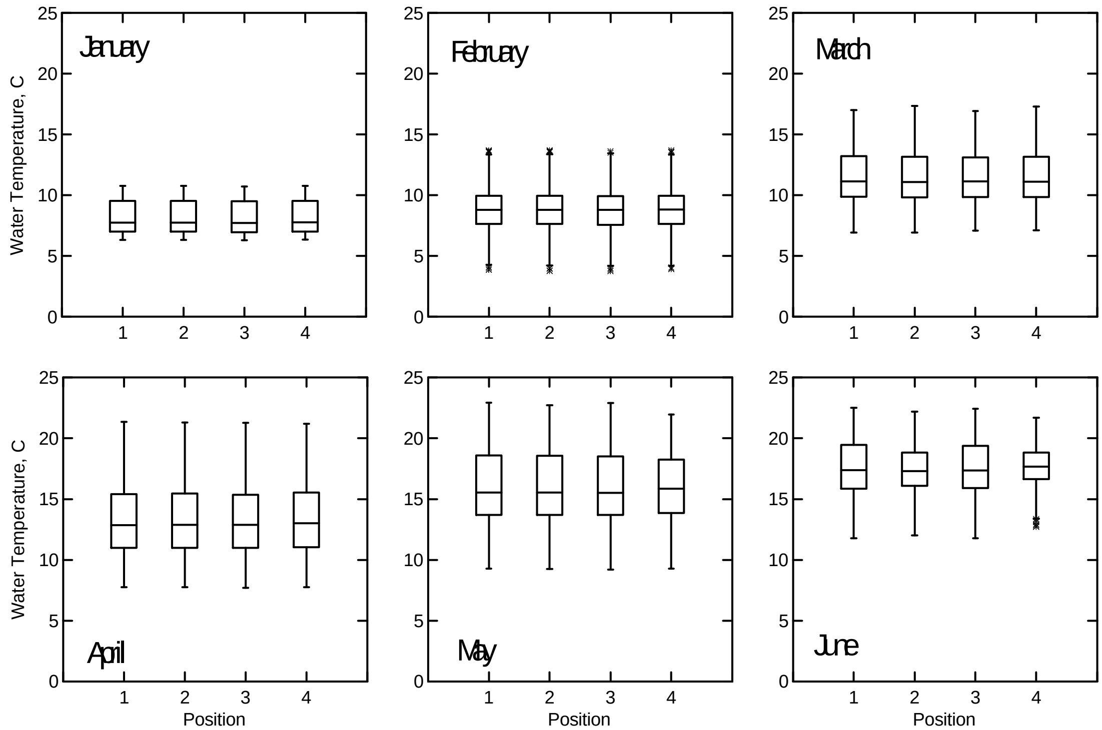


Figure 182. Transect 6 water temperature by logger position and month